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This chapter is one in a series of manuals whose purpose is to provide technical information and policy guidance for the preparation of Waste Load Allocations (WLAs), which are as technically sound as current state of the art permits. The objectives of such load allocations are to ensure that quality conditions that protect designated beneficial uses are achieved. Ad additional benefit of a technically sound WLA is that excessive degrees of treatment, which are neither necessary nor result in corresponding improvement in water quality, can be avoided. This can result in a more effective utilization of available funds.

This chapter addresses Biochemical Oxygen Demand/Dissolved Oxygen (BOD/DO) impacts in streams and rivers.

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Technical Guidance Manual for Performing Waste Load Allocations

Book II
Streams and Rivers

Chapter 1
Biochemical Oxygen Demand / Dissolved Oxygen

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MEMORANDUM

SUBJECT: Technical Guidance Manual for Performing Waste Load Allocations Book II, Streams and Rivers, Chapter 1, BOD/DO

FROM: for Steven Schatzow, Director Albert J. Erickson
Office of Water Regulations and Standards (WH-551)

TO: Regional Water Division Directors
Regional Environmental Services Division Directors
Regional Wasteload Allocation Coordinators

Attached, for national use, is the final version of the Technical Guidance Manual for Performing Waste Load Allocations Book II, Streams and Rivers, Chapter 1, BOD/DO. We are sending extra copies of this manual to the Regional Wasteload Allocation Coordinators for distribution to the States to use in conducting wasteload allocations.

Modifications to the March 1983 draft include:

- Deleting Appendix A and making the Simplified Analytical Method for Determining NPDES Effluent Limitations for POTWs Discharging into Low-Flow Streams available as a separate document.
- Adding a discussion relating the detail of analysis with the anticipated cost of treatment.
- Clarifying the selection of the critical period for the example presented in section 2.
- Warning users against substituting values from the example into real-life situations.
- Adding a note that effluent characteristics used in modeling should reflect the performance expected of the proposed facility during the critical period.

If you have any questions or comments or desire additional information please contact Tim S. Stuart, Chief, Monitoring Branch, Monitoring and Data Support Division (WH-553) on (FTS) 382-7074.

Attachment

TECHNICAL GUIDANCE MANUAL FOR PERFORMING WASTE LOAD ALLOCATIONS

by

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Thomas W. Gallagher contributed to the organization and content of the chapter section dealing with basic concepts and the example analysis.

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SECTION 1.0

INTRODUCTION

1.1 PURPOSE

This chapter is one in a series of manuals whose purpose is to provide technical information and policy guidance for the preparation of Waste Load Allocations (WLAs), which are as technically sound as current state of the art permits. The objectives of such load allocations are to ensure that quality conditions that protect designated beneficial uses are achieved. An additional benefit of a technically sound WLA is that excessive degrees of treatment, which are neither necessary nor result in corresponding improvements in water quality, can be avoided. This can result in a more effective utilization of available funds.

This chapter addresses Biochemical Oxygen Demand/Dissolved Oxygen (BOD/DO) impacts in streams and rivers.

1.2 RELATION TO OTHER BOOKS AND CHAPTERS

Table 1-1 summarizes the relationship of the various "books" and "chapters" that make up the set of technical guidance manuals.

TABLE 1-1

ORGANIZATION OF GUIDANCE MANUAL FOR PERFORMANCE OF WASTE LOAD ALLOCATIONS

BOOK I GENERAL GUIDANCE
(Discussion of overall WLA process, procedures and considerations)

BOOK II STREAMS AND RIVERS
(Specific technical guidance for these water bodies)

Chapter 1 - BOD/Dissolved Oxygen Impacts and Ammonia Toxicity
2 - Nutrient/Eutrophication Impacts
3 - Toxic Substances Impacts

BOOK III ESTUARIES

Chapter 1 - BOD/Dissolved Oxygen Impacts
2 - Nutrient/Eutrophication Impacts
3 - Toxic Substances Impacts

BOOK IV LAKES, RESERVOIRS, AND IMPOUNDMENTS

Chapter 1 - BOD/Dissolved Oxygen Impacts
2 - Nutrient/Eutrophication Impacts
3 - Toxic Substances Impacts

Note: The manual may be expanded to include design conditions, permit averaging periods, screening procedures, and innovative permits. In addition, other water bodies (e.g., groundwaters, bays, and oceans) and other contaminants (coliform bacteria and virus, TDS) may subsequently be incorporated into the manual as the need for comprehensive treatment is determined.

These technical chapters should be used in conjunction with the material in Book I, General Guidance, which provides general information applicable to all types of water bodies and to all contaminants that must be addressed by the WLA process.

Users of this manual should also be aware that other reports and processes may affect the wasteload allocation process. For instance, criteria and standards for dissolved oxygen, ammonia and other parameters are in a continuous process of change. Therefore, any standards used in examples contained in this chapter should not be applied to real-life situations without first consulting the latest applicable criteria and standards documents.

1.3 ORGANIZATION AND SCOPE OF CHAPTER

The remainder of this chapter is organized into three parts, as summarized below.

Section 2.0 presents the underlying technical basis for performing WLAs for the analysis of stream BOD/DO impacts. Both the basic theory and the nature of stream system responses to oxygen demanding loads are described. An example analysis is presented that illustrates the WLA process applied in a simple setting. The object of this section is to provide to technical personnel having limited experience with WLAs, an example that highlights the important issues in performing technically sound analyses and a framework for structuring a consistent approach to more complex problem settings.

Section 3.0 is devoted to a discussion of mathematical models that are required to perform the calculations of water quality impacts on which the WLAs will be based. Guidance is provided to assist in selecting an appropriate model; applying the model to the local situation in a technically sound, consistent manner; and assessing the "goodness" of the final form of the model.

Section 4.0 addresses some key technical considerations that strongly influence the technical adequacy of any analysis performed with the water quality models. The importance of effectively identifying and characterizing water quality problems is discussed, since it will define the focus of the analysis, monitoring programs, and treatment approaches. Data needs are critical and vary with the type of problem and with the model selected. The nature of the data available, even more than the amount of data, will determine the extent to which models can be verified, and the confidence with which WLAs can be established.

1.4 APPROPRIATE LEVELS OF EFFORT IN PERFORMING WLAs

The level of effort that can be applied to the performance of a waste load allocation covers a broad spectrum in terms of resources assigned to collect water quality data and the extent of analysis efforts to calibrate and verify mathematical models. At one extreme, simple preliminary analyses would rely on existing data and estimates of additional information needed to perform the analysis. At the other extreme, WLA studies could be quite thorough and comprehensive.

While an effort approaching either of these extremes could be reasonable and appropriate under a particular set of circumstances, the general case would entail an intermediate level of effort. The degree of confidence desired for identifying the magnitude of water quality impacts for treatment alternatives under consideration will typically require that adequate site-specific data be secured and analyzed. On the other hand, sufficiently reliable engineering analyses can usually be made to support the necessary decisions without the necessity of establishing absolute scientific truth.

The level of effort specified in this manual reflects this intermediate level, which should be sufficient for developing WLAs in the majority of cases. For situations requiring a lesser or greater level of effort, justification should be provided to support such approaches. For many situations involving interacting arrangements of multiple dischargers and complex waste characteristics, the level of effort requires an advanced approach and understanding beyond the scope of this document.

In addition, the complexity of the WLA analysis should also be a function of the anticipated treatment costs; therefore, advanced treatment generally requires more complex WLA analysis including site-specific data collection. However, a level of effort, less than that described in this manual, has been deemed appropriate by EPA for small municipal sewage treatment plants where the discharge is less than 10 MGD and the stream $_7Q_{10}$ low flow is less than the discharge. For these special conditions, the Simplified Analytical Methodology for Determining NPDES Effluent Limitations for POTWs Discharging into Low-Flow Streams (September 26, 1980) and its Addendum (June 25, 1982) can be used.

SECTION 2.0

BASIC PRINCIPLES FOR PERFORMING STREAM BOD/DO WLAs

2.1 GENERAL

Waste Load Allocation (WLA) studies provide information to assist in making effective decisions on levels of treatment required for a source or sources of pollutant load. WLAs are water quality oriented and are directed at establishing a quantitative relationship between a particular waste load and its impact on water quality. These relationships make it possible to compare incremental changes in concentration of specific constituents in the receiving water system. With this capability, one is able to identify the maximum waste load that can be discharged without violating a water quality standard, and to thereby determine a cost-effective level of treatment. Cost-effectiveness, in this sense, relates to the minimum level of treatment that will achieve a specified water quality objective, and assumes that costs are proportional to level of treatment. It should be recognized that further cost-optimization is possible, since a number of treatment system designs exist that produce the same level of treatment at different costs.

A preferred approach to developing cost-effective treatment systems is to use the WLA procedures to define allowable loads. These loads are then compared with existing or projected waste loads to determine the level of control (% removal) required. Alternative treatment approaches are then examined and priced to identify the most effective approach.

The key to making reliable decisions on control requirements necessary to achieve specific water quality objectives lies in the reliability with which waste loads can be related quantitatively to receiving water impacts. These relationships are quite sensitive to natural environmental conditions, such as stream flow and temperature. Because of the variability of such environmental factors, the data available will generally not correspond to all possible conditions (for example, a future population and a critical low-flow condition). Thus, an ability to predict what will happen under the conditions selected for analysis is required.

An additional complication in determining cause-effect relationships and projecting impacts results from the fact that the rate at which various reactions take place is as important as the total amount of waste load generated. This is particularly important in BOD/DO reactions where the resulting dissolved oxygen concentration is determined by competing reactions of oxygen consumption from BOD, ammonia and organic nitrogen decay, and oxygen replenishment from reaeration.

Because of the array of variable elements (temperature, stream flow, load level, reaction rates) that must be considered to establish rate coefficients and examine alternate conditions, computerized mathematical models are generally employed to make the necessary calculations. In the simplest situations (such as employed in the illustrative example at the end of this section), manual calculations can be performed. In most cases, however, the use of computerized mathematical models will be much more convenient. Where the system is relatively complex because of multiple waste load sources, varying stream geometry, flow changes due to tributary inflows, etc., the use of computer models becomes a practical necessity.

One of the disadvantages of using mathematical models is their tendency to prevent the development of an understanding of the system and its responses by most involved parties other than the model analyst. This is because all pertinent interactions are embodied "within the model," and often only the final output is presented for review. However, mathematical models that are properly utilized can contribute greatly to our understanding of the system. This role is emphasized in the discussions in Section 3.0 of this chapter, which addresses the issues in greater technical depth.

2.2 CONCEPTS IN RIVER BOD/DO ANALYSIS

To aid in the general understanding of the nature of river and stream system dissolved oxygen responses to organic waste loads, important relationships are discussed briefly here. An appreciation of the nature and significance of the factors discussed should help an administrator develop a recognition of the significance of certain aspects of the analysis and assist in understanding and evaluating the technical output of a mathematical model analysis.

Transport

When a waste load is discharged into a flowing stream or river, it is subjected to three characteristic factors that tend to modify the concentrations resulting from the initial dilution. The factors that determine the concentration at any particular time or location are:

- Advection - This represents the downstream transport of a discrete element of the waste load by the stream flow.
- Reaction - The biodegradable materials in the waste (BOD) undergo decay under the action of naturally occurring bacteria in the stream. In the presence of dissolved oxygen, bacteria convert the BOD to oxidized end products (e.g., CO_2, NO_3, and H_2O), the result being that the mass of organic matter (BOD) in a discrete element of the waste load gradually diminishes.

- Dispersion - Under the influence of turbulence, eddy currents, and similar mixing forces, a discrete element of the waste load tends not to remain intact, but to mix with adjacent upstream and downstream elements. Dispersion is a predominant factor in tidal waters. In rivers and streams its influence is usually relatively small compared with advection and reaction; however, it can be important in some circumstances. For example, when a slug load results from a spill or accidental dump, dispersion effects can have an important influence on resulting peak concentrations, particularly at longer distances from the point of discharge. Intermittent discharges, such as storm runoff, are also influenced by dispersion. However, for continuous discharges (e.g. from wastewater treatment plants) and steady-state conditions, dispersion effects are usually insignificant, for reasons discussed later in the section.

These factors are shown schematically in Figure 2-1 to illustrate the behavior of a waste load discharged into a stream. A discrete element of the waste load is shown as it is transported downstream. The picture presented is what would be observed if a single slug of waste load were injected and could be followed downstream over a period of time. Conservative constituents in the waste (those not subject to reaction and decay, such as chloride) would track as shown in the sketch for advection only, or advection and dispersion. Reactive constituents, such as BOD, would behave as shown in the sketches that include reaction.

While the sketches represent the behavior of a discrete pulse of waste load, they can be extended to provide a representation of steady-state conditions. Waste load allocations are often performed to examine impacts under a steady-state condition. An extended period of some critical low flow and associated maximum temperature is often selected to represent the design condition. Such conditions normally provide a close enough approximation to a true steady-state to make this type analysis entirely appropriate. For this illustration under steady-state conditions, stream flow and environmental factors affecting reaction rate are constant, and the waste load discharges

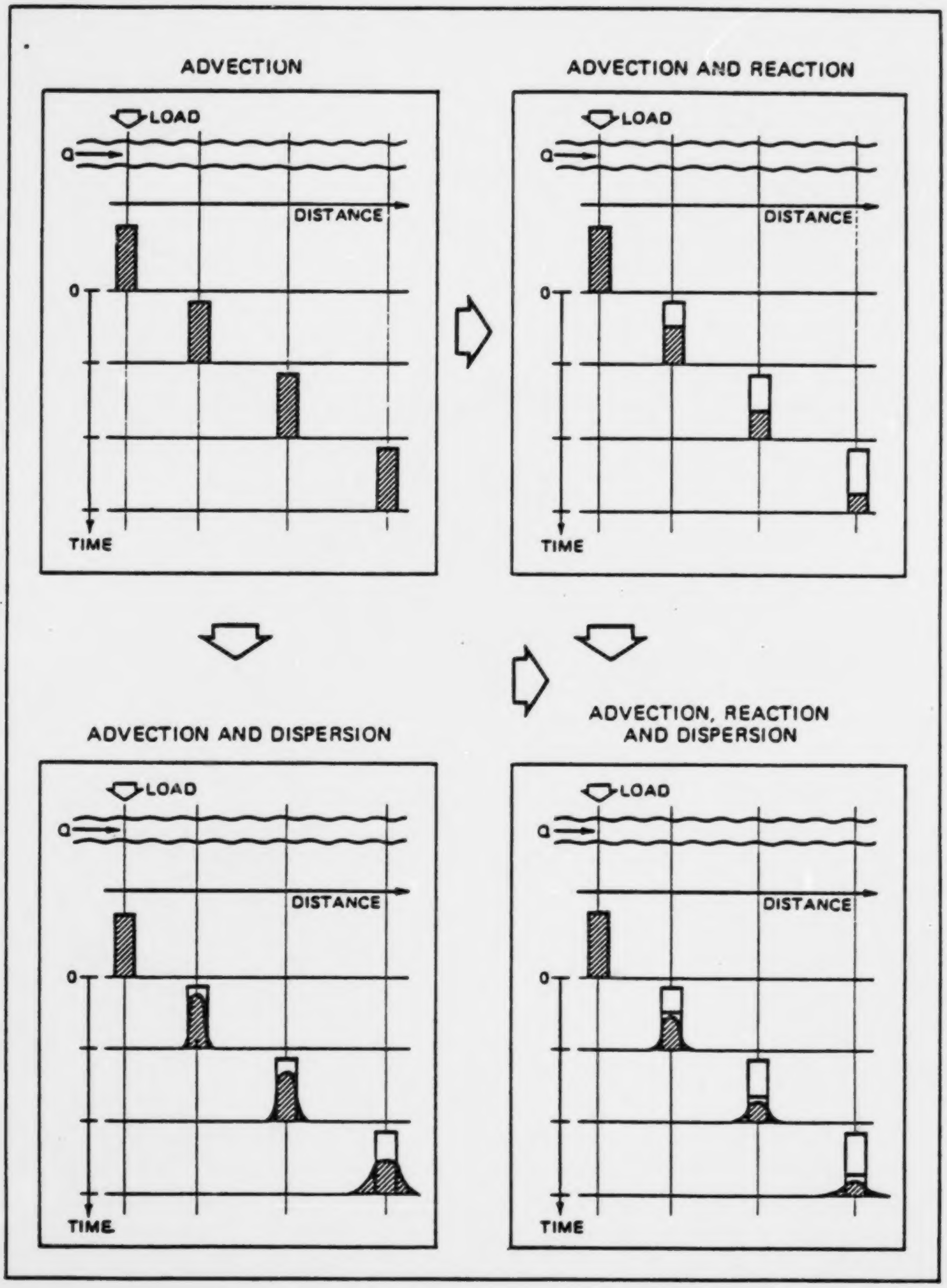
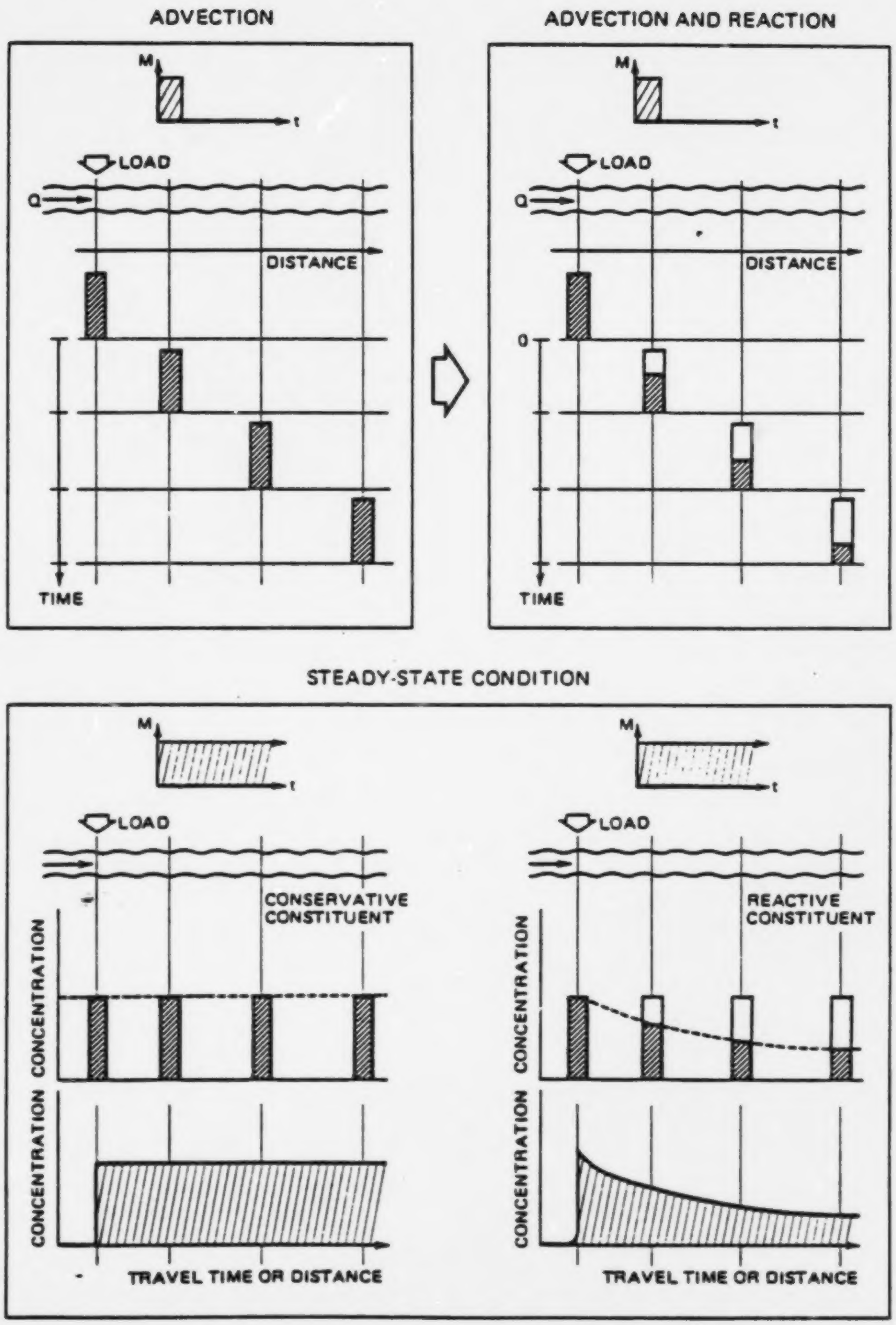

ADVECTION
ADVECTION AND REACTION
ADVECTION AND DISPERSION
ADVECTION, REACTION AND DISPERSION
LOAD
Q
DISTANCE
0
TIME

Figure 2-1. Transport mechanisms for waste loads.

continuously at a constant loading rate. The load pulses shown in Figure 2-1 as different conditions of a single pulse in space and time can also be considered to represent the condition of separate elements of the continuous load being discharged. Under a true steady-state condition, each pulse will behave exactly the same as preceding and following ones. Thus they can be taken (as shown by Figure 2-2) to represent individual points on a continuous profile. Typical concentration profiles are shown for a conservative substance and for a reactive substance such as BOD.

Dispersion has been ignored in these plots. To illustrate why under steady-state conditions, and for conservative constituents it is valid to ignore it in single calculations or why it will not affect results when a computer model which includes dispersion is used, consider Figure 2-3. This presents a set of calculated profiles for a conservative substance (Reaction = 0) under an assumed set of conditions (loading, advection and dispersion). As described earlier, they can represent the concentration profile in the hypothetical stream selected, at successive intervals of 0.1, 0.2, 0.3... days after a load was introduced as a single pulse.

It also represents the group of concentration profiles that would exist in the stream reach shown at any time if the load were introduced in a sequence of pulses spaced 0.1 day apart. At any point along the stream length, the total concentration at that point is made up of components of a number of pulses. By graphically adding up the appropriate individual pulse components, it will be seen that total concentration will be approximately the same at each point in the stream. As pulses are spaced closer together approaching a continuous discharge, the approximation of total concentration at all points will approach the single value represented by W/Q (i.e., mass discharge



ADVECTION

ADVECTION AND REACTION

STEADY-STATE CONDITION

M
t
LOAD
Q
DISTANCE
TIME
0
CONSERVATIVE CONSTITUENT
REACTIVE CONSTITUENT
CONCENTRATION
TRAVEL TIME OR DISTANCE

Figure 2-2. Steady-state responses.

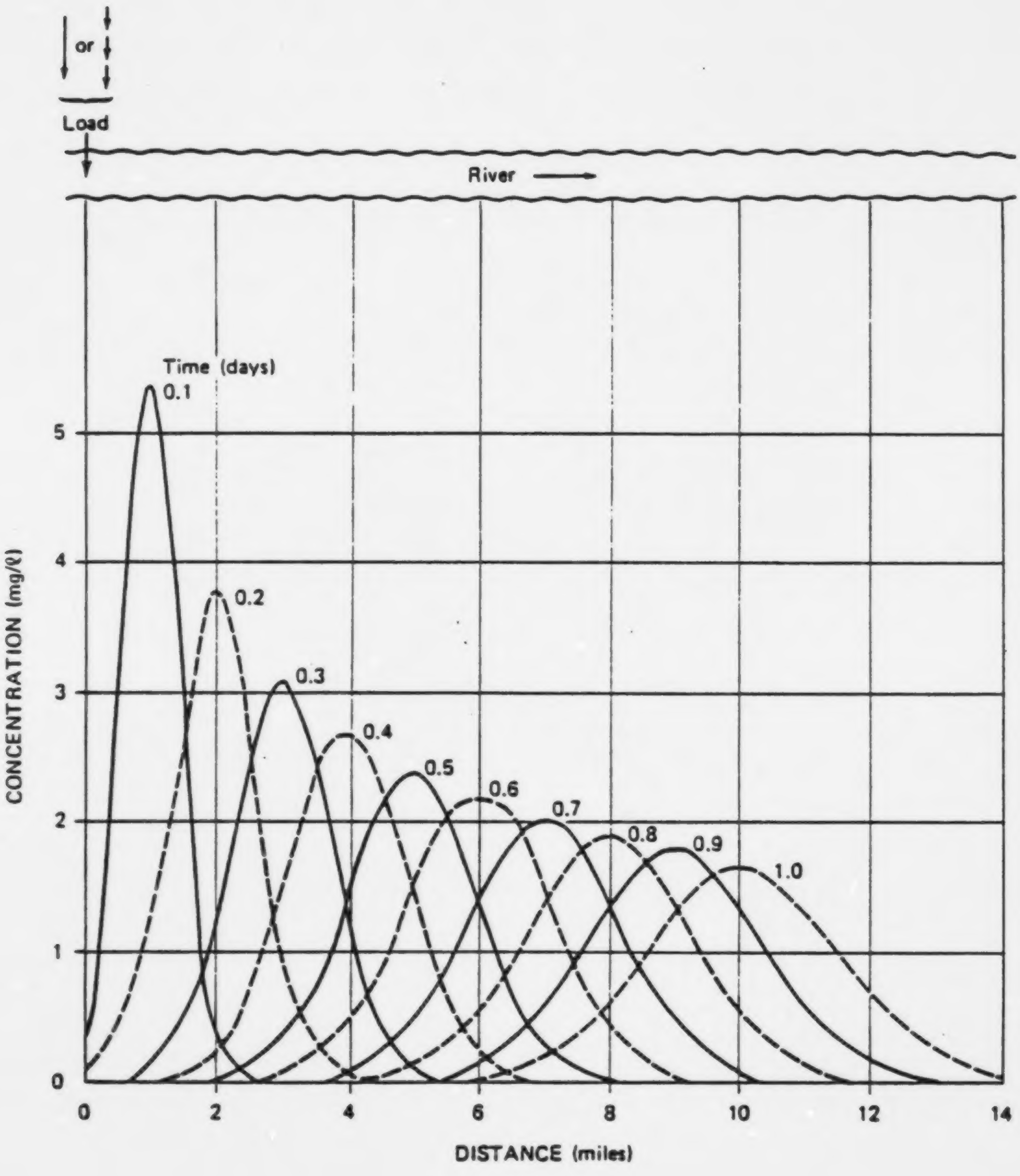


Figure 2-3. Stream dispersion effects.

rate/stream flow rate), shown as the steady-state profile for a conservative constituent in the lower left plot in Figure 2-2.

This is true for conservative constituents, and approximately for reactive ones. For most advective streams, the impact of longitudinal dispersion on the profile of a reactive substance can be ignored. There are exceptions, however, and procedures for estimating the magnitude of dispersive forces and their effect on water quality impacts are presented in section 3.1.

BOD Reactions

BOD (biochemical oxygen demand) is a measure of biodegradable material in terms of the oxygen utilized in stabilizing it. Both carbonaceous organic compounds (CBOD) and nitrogenous forms (NBOD), principally ammonia and organic nitrogen, are subject to bio-oxidation. For convenience, the oxygen equivalent of biodegradation over a 5-day period is usually measured (BOD_5) rather than the full oxygen equivalency (Ultimate BOD, ULT BOD, BOD_u), which commonly requires in excess of twenty or thirty days for completion. In some cases, such as with pulp and paper mill effluents, the BOD_u test can require over one hundred days. The rate at which biodegradable material (CBOD or NBOD) is removed in a stream may be determined from an analysis of river BOD_5 or BOD_u, since both are suitable indicators of biodegradable material present. The analysis of BOD (whether ultimate or 5-day) referred to in this manual utilizes a nitrification-inhibited test, unless stated otherwise; thus ratios of ultimate oxygen demand to 5-day oxygen demand are for carbonaceous demands only. Table 3-20 gives a range of values appropriate for the $CBOD_u/CBOD_5$ ratio.

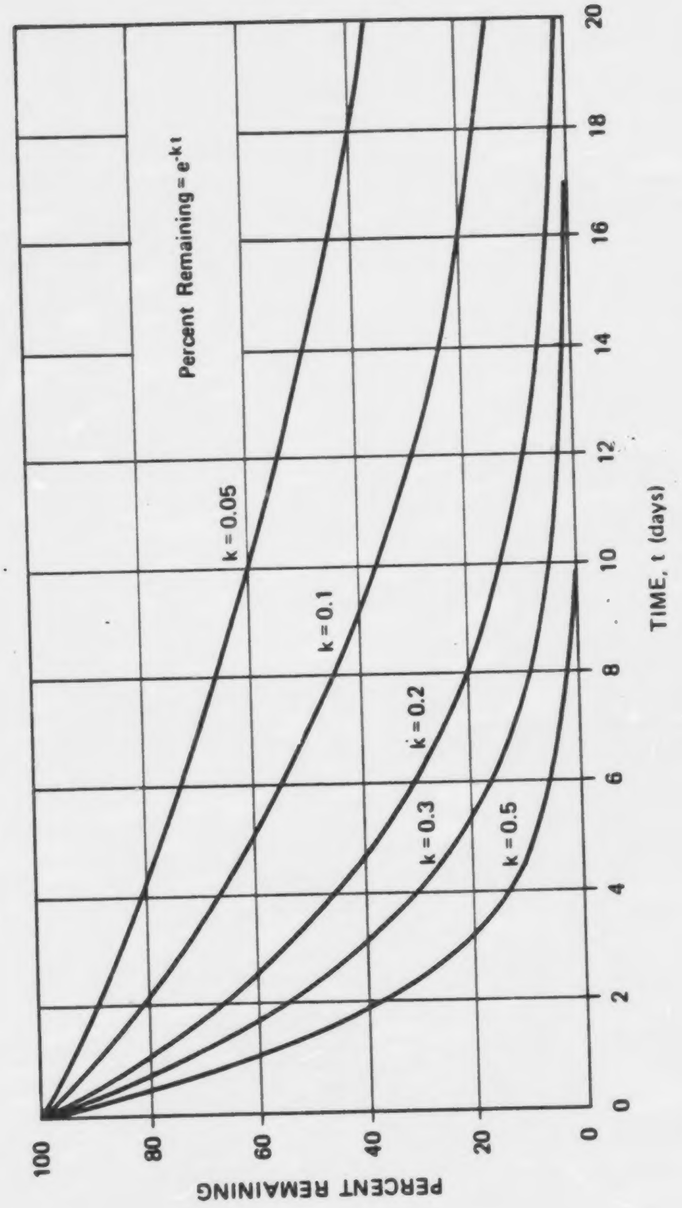
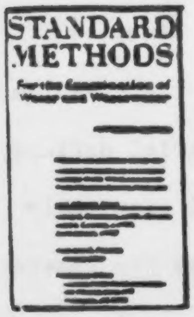


Figure 2-4. BOD decay: range of reaction rates.

Table 2-1. Solubility of Oxygen in Water Exposed to Water-Saturated Air (mg/l)**

Temperature C	Chloride Concentration in Water mg/L: 0	5.000	10.000	15.000	20.000
0	14.60	13.72	12.90	12.13	11.41
1	14.19	13.35	12.56	11.81	11.11
2	13.81	12.99	12.23	11.51	10.83
3	13.44	12.65	11.91	11.22	10.56
4	13.09	12.33	11.61	10.94	10.30
5	12.75	12.02	11.32	10.67	10.05
6	12.43	11.72	11.05	10.41	9.82
7	12.12	11.43	10.78	10.17	9.59
8	11.83	11.16	10.53	9.93	9.37
9	11.55	10.90	10.29	9.71	9.16
10	11.27	10.65	10.05	9.49	8.96
11	11.01	10.40	9.83	9.28	8.77
12	10.76	10.17	9.61	9.08	8.58
13	10.52	9.95	9.41	8.89	8.41
14	10.29	9.73	9.21	8.71	8.24
15	10.07	9.53	9.01	8.53	8.07
16	9.85	9.33	8.83	8.36	7.91
17	9.65	9.14	8.65	8.19	7.78
18	9.45	8.95	8.48	8.03	7.61
19	9.26	8.77	8.32	7.88	7.47
20	9.07	8.60	8.16	7.73	7.33
21	8.90	8.44	8.00	7.59	7.20
22	8.72	8.28	7.85	7.45	7.07
23	8.56	8.12	7.71	7.32	6.95
24	8.40	7.97	7.57	7.19	6.83
25	8.24	7.83	7.44	7.06	6.71
26	8.09	7.69	7.31	6.94	6.60
27	7.95	7.55	7.18	6.83	6.49
28	7.81	7.42	7.06	6.71	6.38
29	7.67	7.30	6.94	6.60	6.28
30	7.54	7.17	6.83	6.49	6.18
31	7.41	7.05	6.71	6.39	6.08
32	7.28	6.94	6.61	6.29	5.99
33	7.16	6.82	6.50	6.19	5.90
34	7.05	6.71	6.40	6.10	5.81
35	6.93	6.61	6.30	6.01	5.72
36	6.82	6.51	6.20	5.92	5.64
37	6.71	6.40	6.11	5.83	5.56
38	6.61	6.31	6.02	5.74	5.48
39	6.51	6.21	5.93	5.66	5.40
40	6.41	6.12	5.84	5.58	5.33
41	6.31	6.03	5.76	5.50	5.25
42	6.22	5.94	5.68	5.42	5.18
43	6.13	5.85	5.60	5.35	5.11
44	6.04	5.77	5.52	5.27	5.04
45	5.95	5.69	5.44	5.20	4.98
46	5.86	5.61	5.37	5.13	4.91
47	5.78	5.53	5.29	5.06	4.85
48	5.70	5.45	5.22	5.00	4.78
49	5.62	5.38	5.15	4.93	4.72
50	5.54	5.31	5.08	4.87	4.66



** At a total pressure of 101.3 kPa. Under any other barometric pressure, *P*, obtain the solubility, *S'* (mg/L) from the corresponding value in the table by the equation:

$$S' = S\,\frac{P-p}{760-p}$$

in which *S* is the solubility at 101.3 kPa and *p* is the pressure (mm) of saturated water vapor at the water temperature. For elevations less than 1,000 m and temperatures below 25 C, ignore *p*. The equation then becomes:

$$S' = S\,\frac{P}{760} = S\,\frac{P'}{29.92}$$

Dry air is assumed to contain 20.90% oxygen. (Calculations made by Whipple and Whipple, 1911. *J. Amer. Chem. Soc.* 33:362.)

of water temperature on saturation concentrations is the factor of major practical significance. Table 2-1 summarizes the effects of salinity and temperature. Freshwater streams, which can experience temperatures between 0 and about 30°C, will have dissolved oxygen saturation concentrations between about 15 and 7.5 mg/l, respectively.

Figure 2-5 illustrates the effect of altitude on the saturation concentration of dissolved oxygen. Saturation values for DO in rivers and streams at elevations of 4000 to 5000 feet will be about 1 to 1.5 mg/l lower than in lowland streams at similar temperatures.

A stream's ability to exhibit self-purification is related to the ability of naturally occurring bacteria to decompose the organic waste materials, utilizing the oxygen resources of the stream, coupled with the ability of the stream to replenish these resources by natural reaeration processes. Transfer of atmospheric oxygen to the water column occurs through diffusion and turbulence. Of critical importance to the protection of water quality—one aspect which is usually defined in terms of a minimum acceptable concentration for dissolved oxygen—is the rate at which reaeration takes place and the magnitude of this rate in relation to the rate of oxygen consumption.

Most analytical methods are based on the concept of oxygen deficit (D), defined as the difference in concentration between the saturation value (C_S) and the actual DO concentration (C).

Like BOD, reaeration is considered to follow "first-order" kinetics, such that the rate of reaeration at any time is proportional to the dissolved oxygen deficit at that time. An equation of fraction of the initial deficit remaining versus time would have the same form as that presented previously for BOD decay, except that the reaction rate coefficient would be the reaeration

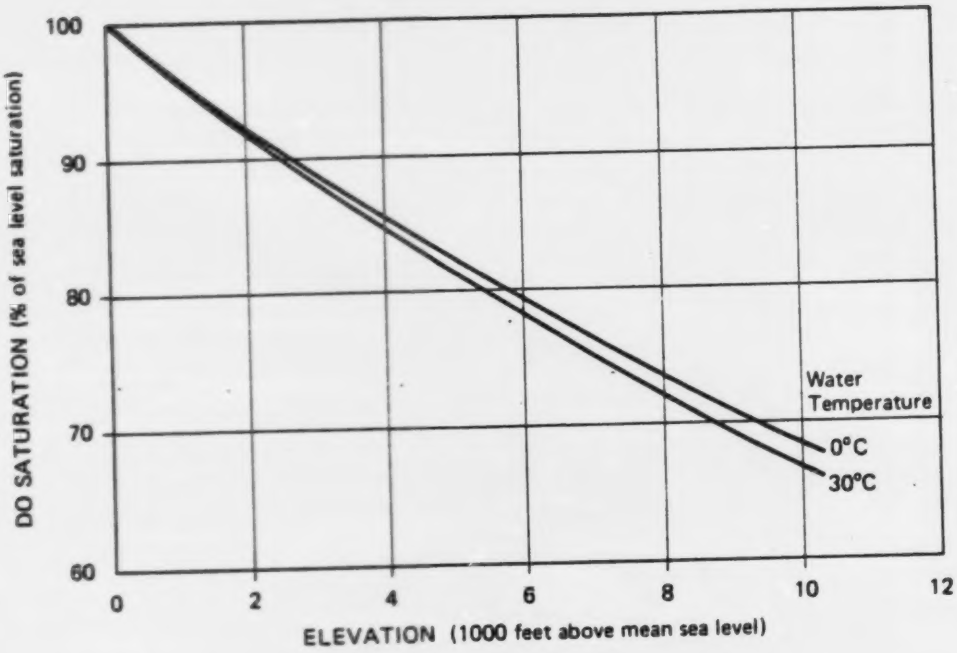


Figure 2-5. Effect of elevation on dissolved oxygen saturation concentration.

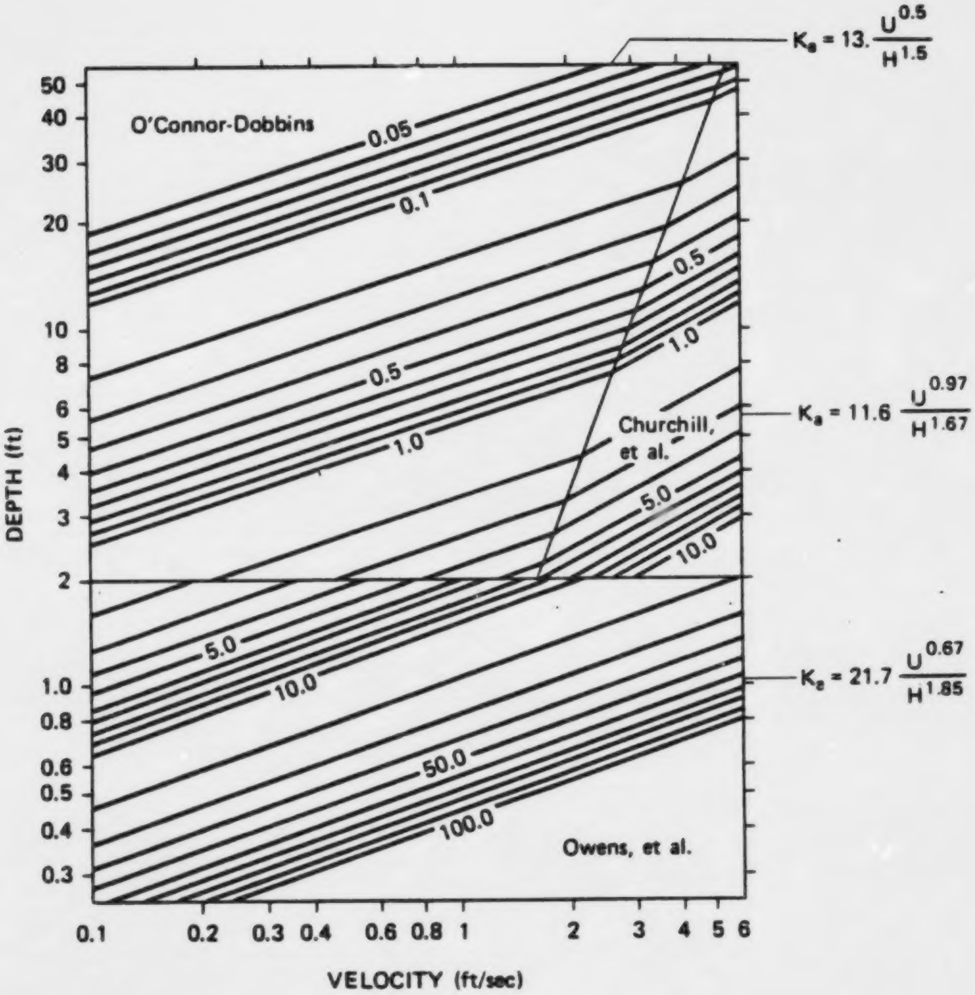
rate. In an artificial environment, that is, starting at zero dissolved oxygen, and with no significant influences other than reaeration, a plot of fraction of the initial deficit remaining versus time would resemble the BOD plot shown in Figure 2-4.

Reaeration rate coefficients, however, span a wider range of values and have a greater magnitude (typically 0.1 to 5.0 or even greater) than BOD exertion rates. While influenced by temperature, they are most strongly correlated with depth and velocity of the stream. Figure 2-6 summarizes the influence of velocity and depth on reaeration rates. This figure associates different reaeration formulas at the regimes of depth and velocity which were used to develop the individual formulations. Alternatively, the reaeration rate can be measured for each reach. See also the part of Section 3.3 which deals with the reaeration rate.

Dissolved Oxygen Profile

In natural waters, the oxygen consumed by bacteria in oxidizing the biodegradable organic matter in a wastewater discharge (BOD) is taken from the dissolved oxygen originally present in the water and from the additional amounts transferred into the water by atmospheric reaeration. This is illustrated graphically in Figure 2-7.

The top figure is a calculated BOD profile in a river with a BOD removal rate of 0.5 per day. At time t = 0, there is in this example, a concentration of 10 mg/l present, and after about 10 days all of the biochemical oxygen demand (BOD) has been exerted. Since the BOD test measures the amount of organic matter present directly in terms of the amount of oxygen required for its stabilization by biological action, the reduction in BOD concentration is equivalent to dissolved oxygen consumption.



Cited in EPA-600/3-78-105 "Rates, Constants, and Kinetic Formulations in Surface Water Quality Modeling" (2) in modified form.

Figure 2-6. Reaeration rate coefficient (K_a) related to depth and velocity.

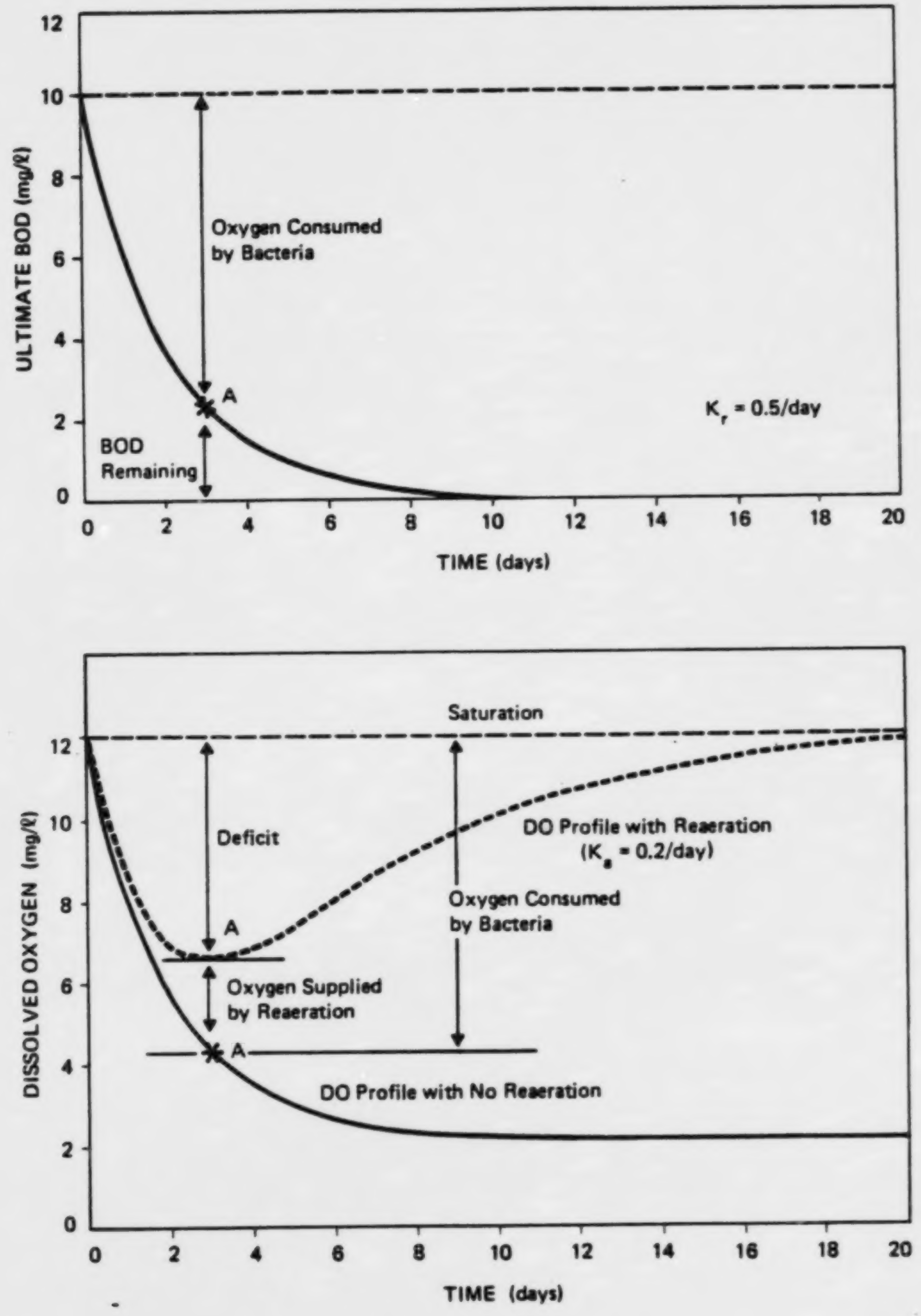


Figure 2-7. Stream reaeration relationships.

The bottom figure shows two calculated dissolved oxygen profiles associated with the BOD removal profile in the upper plot. The profile indicated by the solid line represents conditions that would occur in a river if oxygen were not replenished by reaeration. In this case, the assumed initial dissolved oxygen concentration of 12 mg/l is ultimately reduced to 2 mg/l to satisfy the ultimate BOD of 10 mg/l. The dotted profile illustrates the net effect of reaeration providing an additional source of oxygen.

The characteristic shape of the stream dissolved oxygen profile (the DO sag curve) is the result of the interplay of the oxidation and reaeration reaction rates. Each is represented by first-order kinetics: the rate of oxygen consumed is proportional to the concentration of BOD remaining at any time, and the rate of oxygen supplied is proportional to the magnitude of the deficit at any time.

In the early stages, oxidation greatly exceeds reaeration because BOD concentrations are high and river dissolved oxygen concentrations are close to saturation (i.e., deficits are small). Oxygen is used faster than it is replaced, and stream dissolved oxygen concentrations decrease. As time progresses, the consumption of oxygen decreases as the amount of BOD remaining is reduced, and the supply of oxygen increases as stream concentrations drop and deficits become greater. At some point the decreasing utilization and the increasing supply are equal since oxygen is supplied at the same rate it is utilized. This situation defines the "critical" point when the lowest concentration of dissolved oxygen will be reached in the stream. Although the rate of supply gradually diminishes after this, it always exceeds the utilization rate, which continues to drop. River dissolved oxygen concentrations increase thereafter, though at a decreasing rate as concentrations approach saturation. Figure 2-8 presents a set of computations performed to illustrate the nature of stream system responses under the influence of a range of reaction rate

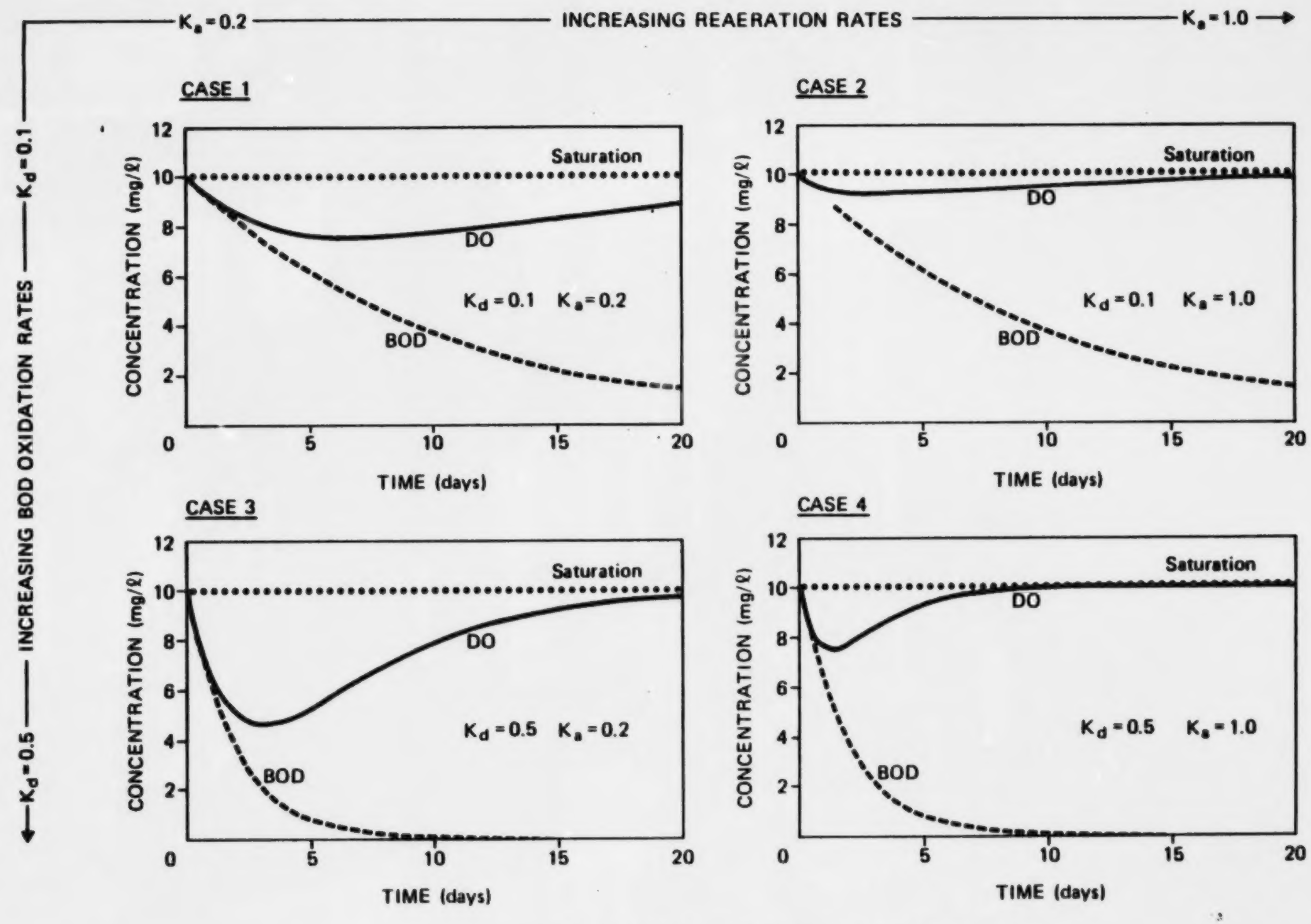


Figure 2-8. Effect of rates on stream DO impacts.

values for BOD oxidation (D_d) and reaeration rates (K_a). For simplicity, in these examples the initial deficit (D_o) is assumed to be zero; that is, the stream is at saturation concentration (assumed to be 10 mg/l) after initial mixing of the waste load. It is also assumed that all BOD removal occurs through oxidation. The initial BOD concentration (L_o) is assumed to be 10 mg/l; corresponding, for example, to a BOD load of 10,000 lbs/day discharged into a stream with a flow of 185 cfs.

Figure 2-8 presents a range of stream DO responses that would result from various combinations of high and low values for these rate coefficients. For any given BOD waste load (and hence initial stream concentration), both the magnitude of the maximum DO depression, and the location in the stream at which it occurs will be determined by the particular combination of rate coefficients which apply for that system. The rate coefficients also affect the total length of the river over which impact is detected.

Shallow, swift-flowing streams tend to have higher values of both K_a and K_d, and tend to be represented by Case 4 in Figure 2-8. In contrast, deep, slow-moving rivers tend to have lower values of K_d and K_a, as illustrated by Case 1. In such streams the reaction is spread over a longer time, and usually a correspondingly longer distance, and the DO sag becomes less pronounced, making it more difficult to readily observe the effect of the waste discharge from stream water quality data because of the potential to be masked by reactions from other loads or sources occurring farther downstream (refer also to the parts of Section 3.3 dealing with K_a and K_d).

The dissolved oxygen values plotted were obtained by subtracting the calculated deficit concentration from the saturation concentration (in the example assumed to be 10 mg/l, corresponding to a stream temperature of about 15°C). If the saturation value was 9 mg/l (stream temperature about 20°C), then each point on the dissolved oxygen curves would be 1 mg/l lower. The principal effect of temperature is on dissolved oxygen saturation values, but temperature also affects reaction rates. Higher temperatures increase both BOD exertion and reaeration rates. The effect on the BOD exertion rate is more pronounced, so that higher stream temperatures produce a net decrease in dissolved oxygen.

Variations in River Velocity, Depth, Area with Flow

Projections of water quality impacts for some future critical low-flow condition are normally required in waste load allocation studies. The predominant impact of reduced stream flow is usually to reduce dilution provided for the waste load. Thus initial concentrations of BOD (L_o) become significantly higher. However, reduced stream flows also result in changes in stream velocity and depth--factors which both strongly affect the reaeration rate coefficient. Therefore, the effect of stream flow changes on depth and velocity must be determined.

In some cases stream cross-sectional measurements and time of passage (travel time) information will be available from studies performed by USGS or state environmental agencies. In addition, data may be available from the Corps of Engineers in locations where projects have been designed. Time of passage data can be used to compute averages of velocity (U) and depth (H). Each set of data will be related to a specific stream flow regime, and different values can be expected for other flow regimes. In seeking information on depth/velocity relationships in a stream reach, it is important to distinguish between time of travel study data and information that might be derived from USGS flow gaging stations. Gaging stations are usually located at places where the geometry favors the accurate determination of flow rate, and as a result, depth and velocity at such locations are not typical of general stream conditions.

Leopold and Maddox (3) have suggested the following empirical relationship between the pertinent physical stream factors and stream flow.

$$U = aQ^n \tag{2-3}$$

$$H = bQ^m \tag{2-4}$$

$$W = cQ^f \tag{2-5}$$

where:

a, b, c are constants for the stream in question

m, n, f are exponents defining the basic relationships

Recognizing that stream flow is the product of cross-sectional area and velocity ($Q = A \cdot U$), and that cross-sectional area is the product of width and depth ($A = W \cdot H$), it can be shown that the sum of the exponents ($m + n + f$) is 1. Using this and experience from a variety of streams, the value of the exponents can be approximated as follows:

n = range (0.4 - 0.6); typical 0.5

m = range (0.3 - 0.5); typical 0.4

f = range (0 - 0.2); typical 0.1

Where the analyst has more than one set of data, a log-log plot of area (A), depth (H) and velocity (U) against stream flow (Q) will permit extrapolation to other flows of interest. The slope of such plots provides the local value of the exponents. When data at only a single flow regime is available, estimates for other flows of interest can be developed by the following ratios, derived from the foregoing relationships:

velocity : $U_2/U_1 = (Q_2/Q_1)^{0.5}$ (2-6)

depth : $H_2/H_1 = (Q_2/Q_1)^{0.4}$ (2-7)

cross-sect. area : $A_2/A_1 = (Q_2/Q_1)^{0.5}$ (2-8)

travel time : $T_2/T_1 = (Q_2/Q_1)^{-0.5}$ (2-9)

It should be recognized that these relationships exist only in free-flowing streams, and that the exponents may vary by 50% for any river. Impounded reaches in rivers have exponents m and $f=0$, and $n=1$. Thus, acquiring data to develop site-specific relationships of this type is normally appropriate.

Because of the nature of the relationships, reduced stream flows tend to result in increased reaeration rates, principally due to the beneficial effect of shallower depths. For example, a 50% reduction in stream flow may increase K_a by about 30%. However, the net effect on stream DO impacts is negative because the improved reaeration is outweighted by increases in initial BOD concentations (L_o), which would double for a 50% reduction in stream flow.

2.3 DEVELOPMENT OF BOD/DO EQUATIONS FOR RIVERS

The principle upon which the equations in all water quality models are based is a simple mass balance: mass leaving a river segment is equal to mass entering the segment plus mass added directly to the segment, less any mass lost within the segment. No matter how complicated the final water quality equations appear, they are all a mathematical statement of this mass balance principle. The applications of the mass balance principle to BOD and dissolved oxygen in streams follows. For instructive purposes, more detail is given in the development of the BOD equation.

BOD. To develop the basic BOD equation, consider a small segment of a stream at any arbitrary location (x) along its length, as illustrated by Figure 2-9. A BOD mass balance can be made on the small segment of length Δx as follows. At steady state, the following mass balance applies:

$$\text{MASS OUT} = \text{MASS IN} - \text{LOSS}$$

$$Q\left(L + \frac{dL}{dx}\Delta x\right) = QL - K_r LV$$

$$QL + Q\frac{dL}{dx}\Delta x = QL - K_r LV \qquad (2\text{-}10)$$

note that: $U = Q/A$; $V = A\Delta x$

and therefore $Q\Delta x\frac{dL}{dx} = UA\Delta x\frac{dL}{dx} = UV\frac{dL}{dx}$.

divide 2-10 by volume V:

$$\frac{QL}{V} + \frac{UVdL}{V\,dx} = \frac{QL}{V} - \frac{K_r LV}{V}$$

dividing and transposing, the equation reduces to:

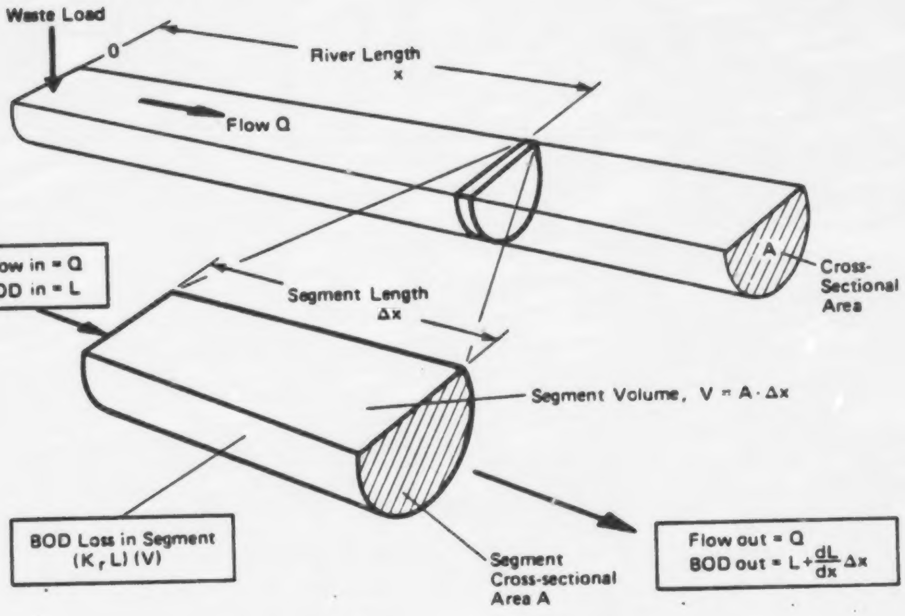
$$U\frac{dL}{dx} + K_r L = 0 \qquad (2\text{-}11)$$

integration of this equation yields

$$L = C\,e^{\frac{-K_r x}{U}} \qquad (2\text{-}12)$$

where C is the constant of integration.

The value of the constant of integration C can be determined by applying the equation at the point of discharge of the waste load into the stream, i.e., at $x = 0$. At this point the BOD concentration (L) is equal to the initial


Waste Load
0
River Length
x
Flow Q
A
Cross-Sectional Area
Flow in = Q
BOD in = L
Segment Length
Δx
Segment Volume, V = A · Δx
BOD Loss in Segment
(K$_r$ L) (V)
Segment Cross-sectional Area A
Flow out = Q
BOD out = L + dL/dx Δx

Q = River flow rate

L = BOD concentration entering segment; i.e. concentration at location x

QL = BOD mass entering segment

$\frac{dL}{dx}$ = Rate of change of BOD (L) with distance (x); equivalent to rate of change with time (t) when converted by velocity = (U); thus $U = \frac{Q}{A}$; $t = \frac{x}{U}$

$\frac{dL}{dx}\Delta x$ = Change in BOD concentration during time of passage through segment of length Δx

$L + \frac{dL}{dx}\Delta x$ = BOD concentration leaving segment; i.e. concentration at location $x + \Delta x$

K_r = BOD removal rate coefficient; reflects first-order removal of BOD according to the relationship $e^{-K_r t} = e^{-K_r x/U}$ = fraction of BOD removed

$K_r L$ = Change in BOD concentration in segment

$K_r LV$ = Change in BOD mass in segment

Figure 2-9. Basic relationships for BOD equations.

concentration of BOD immediately after dilution by the stream flow. This initial concentration is designated L_o. Thus,

$$L = L_o \text{ at } x = 0$$

substituting L_o for L and 0 for x in equation (2-12) yields

$$L_o = C e^{\frac{-K_r x}{U}} = C e^{\circ} = C(1)$$

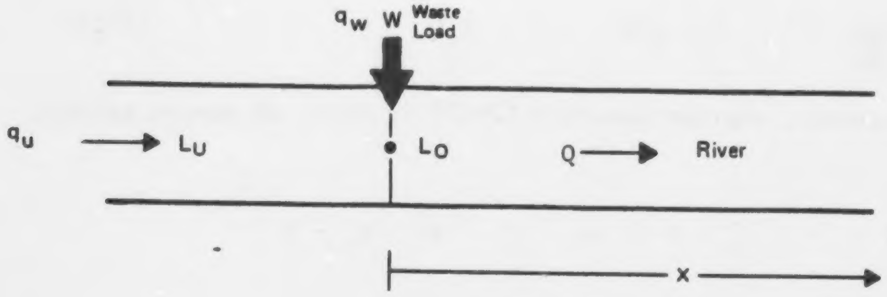
$$L_o = C$$

substituting for C in equation (2-12)

$$L = L_o e^{\frac{-K_r x}{U}} \qquad (2\text{-}13)$$

Equation (2-13) provides the means for calculating the BOD concentration (L) at any stream location (x) under the influence of advective transport (defined by velocity, U) and the removal reaction (represented by rate coefficient, K_r). It can be applied for any substance that oxidizes or is removed at first-order rates. Such reactions include oxidation of BOD and ammonia, or the die-off of coliform bacteria.

The value of the initial BOD concentration (L_o) can be derived from a simple mass balance. Where the upstream flow (q_u) has a significant concentration (L_u), it is incorporated in the calculation.



$$L_o = \frac{L_u q_u + W}{Q}$$

$$Q = q_u + q_w$$

In the above development, the rate of BOD removal was defined by the parameter (K_r). However, in a natural body of water, the rate of BOD removal does not necessarily equal the rate at which oxygen is utilized to satisfy the BOD. This parameter (the rate of BOD oxidation) is defined as (K_d). The parameter (K_d) defining oxidation of BOD can be equal to or less than the parameter (K_r) which defines total removal. The difference reflects removal by physical processes, settling for example.

Dissolved Oxygen. The dissolved oxygen equation is developed with the same procedure used for the BOD equation. The basic elements for the DO equation development are illustrated in Figure 2-10. A dissolved oxygen mass balance can be made on the small segment of length Δx, as follows. At steady state, the following mass balance applies.

$$\text{MASS IN} - \text{MASS OUT} + \text{SOURCES} - \text{SINKS} = 0$$

$$QC - Q\left(C + \frac{dC}{dx}\Delta x\right) + K_a(C_s - C)V - K_d LV = 0$$

$$-\frac{QdC}{dx}\Delta x + K_a(C_s - C)V - K_d LV = 0 \quad (2\text{-}14)$$

Note that $U = Q/A$; $V = A\Delta x$

$$\text{and therefore} \quad \frac{QdC\Delta x}{dx} = (Q\Delta x)\frac{dC}{dx} = (UA\,\Delta x)\frac{dC}{dx} = UV\frac{dC}{dx}$$

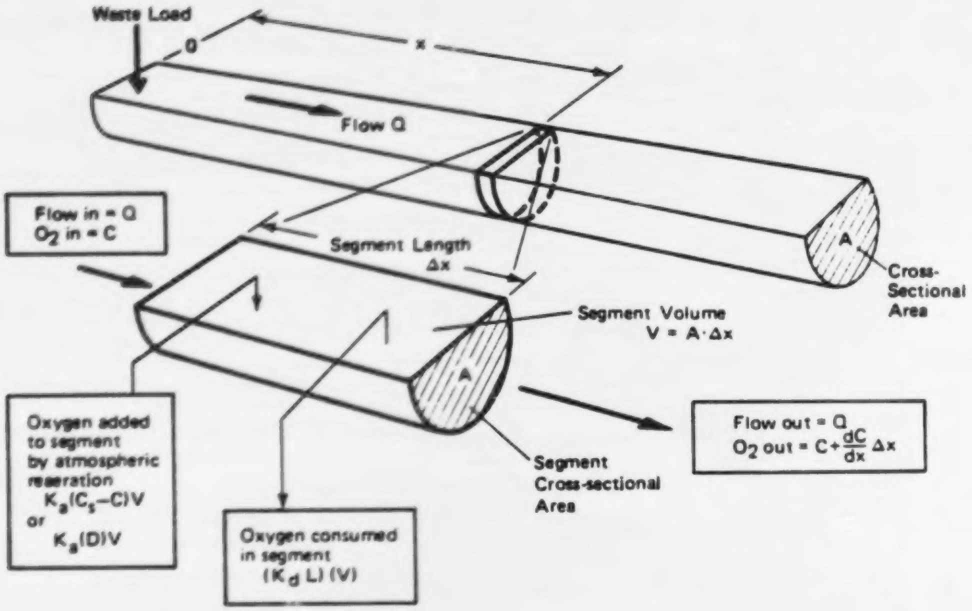
Divide equation 2-14 by volume V

$$-\frac{UdC}{dx} + K_a(C_s - C) - K_d L = 0 \quad (2\text{-}15)$$

For mathematical convenience, express equation (2-15) in terms of oxygen deficit (D), where:

$$D = C_s - C \quad \text{or} \quad C = C_s - D$$

$$-\frac{Ud(C_s - D)}{dx} + K_a D - K_d L = 0 \quad (2\text{-}16)$$


Waste Load
0
x
Flow Q
A
Cross-Sectional Area
Flow in = Q
O_2 in = C
Segment Length Δx
Segment Volume $V = A \cdot \Delta x$
A
Flow out = Q
O_2 out = $C + \frac{dC}{dx} \Delta x$
Segment Cross-sectional Area
Oxygen added to segment by atmospheric reaeration $K_a(C_s - C)V$ or $K_a(D)V$
Oxygen consumed in segment $(K_d L)(V)$

Q = River flow rate

C = Concentration of dissolved oxygen entering segment; i.e. concentration at location x

C_s = Saturation concentration of dissolved oxygen

QC = Mass of oxygen entering segment

$\frac{dC}{dx}$ = Rate of change of oxygen (C) with distance (x); equivalent to rate of change with time (t) when converted by velocity (U)

$\frac{dC}{dx} \Delta x$ = Change in oxygen concentration during time of passage through segment of length Δx

$C + \frac{dC}{dx} \Delta x$ = Dissolved oxygen concentration leaving segment; i.e. at location $(x + \Delta x)$. This term multiplied by flow (Q) yields mass of O_2 leaving segment.

K_a = Atmospheric reaeration rate; reflects first-order reaction whereby fraction of oxygen deficit satisfied = $e^{-K_a t} = e^{-K_a x/U}$

$K_a D$ or $K_a(C_s - C)$ = Change in dissolved oxygen concentration in segment; this term, multiplied by segment volume (V) yields change in dissolved oxygen mass in segment.

K_d = BOD oxidation rate; where oxidation accounts for all BOD removal, $K_d = K_r$ used in development of BOD equations.

Figure 2-10. Basic relationships for DO equations.

This can also be written

$$- U\left[\frac{dC_s}{dx} - \frac{dD}{dx}\right] + K_aD - K_dL = 0 \qquad (2\text{-}17)$$

When C_s is constant with respect to x;

$$\frac{dC_s}{dx} = 0$$

and the expression reduces to

$$+ U\frac{dD}{dx} + K_aD - K_dL = 0 \qquad (2\text{-}18)$$

Substitute for L the expression in equation 2-13 and rearrange terms

$$\frac{UdD}{dx} + K_aD = K_dL_oe^{-K_r x/U} \qquad (2\text{-}19)$$

Integrating and solving equation (2-19) for the condition that $D = D_o$ at $x = 0$, yields:

$$D = D_oe^{-K_a x/U} + \frac{K_d}{K_a-K_r} L_o\left[e^{-K_r x/U} - e^{-K_a x/U}\right] \qquad (2\text{-}20)$$

2.4 EXAMPLE WASTE LOAD ALLOCATION ANALYSIS

The following waste load allocation analysis is designed to provide an example that is reasonably realistic but not so complex that it detracts from the example's principal intent, which is to:

- demonstrate the major steps in a waste load allocation analysis,
- show how the mathematical equations presented in previous sections are applied, and
- present the relationship between water quality impacts and the overall task of performing a waste load allocation.

Readers are cautioned that site-specific data must be used when performing wasteload allocation analyses, and that values presented in this (or any other) example must not be substituted for site-specific data. When such data is not available, applicable values should be developed by following the procedures detailed in the text. Because the purpose of this example is to present an overview of the steps required for a waste load allocation, emphasis is not placed on providing details on data requirements and calibration-validation procedures. These technical aspects are discussed more thoroughly in other sections of the manual.

Problem Setting. In this example, a city of approximately 60,000 people discharges its wastewater into a relatively small river with an average annual flow of about 250 cfs. The city's wastewater is presently treated by a trickling filter plant that provides about 85% BOD removal and has reached its design capacity of 7.5 MGD. The population is projected to increase by more than 50% to 92,000 people (with a range of 75,000 to 120,000 people) by the year 2000. Expansion of the treatment plant to a capacity of 11.5 MGD and provision of an activated sludge system for secondary treatment are proposed.

The river, for thirty miles downstream of the treatment plant discharge, is classified as B1, which has a designated water use for fish and wildlife propagation. The pertinent State water quality standards for this example are a minimum dissolved oxygen level of 5.0 mg/1 and a maximum un-ionized ammonia concentration of 0.02 mg/1. The river is used locally for fishing and is bordered by several campgrounds and a state park. No water quality problems are documented, and the limited water quality data do not show any standards violations. A summary of the problem setting and treatment plant data is presented in Figures 2-11 and 2-12.

River Characteristics. The river flow is gauged by the USGS immediately upstream of the treatment plant discharge. The average monthly flows for a thirty-year period are summarized in Figure 2-13. The average annual flow is about 250 cfs with a minimum monthly average low flow of 100 cfs, which occurs in September. However, the State requires that minimum dissolved oxygen standards must be met for a minimum seven-consecutive-day flow with a return period of once met for a minimum seven-consecutive-flow with a return period of once every 10 years ($_7Q_{10}$). (As discussed in Book VI, Design Conditions other design flows may apply under summer conditions.) From a statistical analysis of the flow records, the $_7Q_{10}$ is determined to be 30 cfs and occurs between August and October (for further discussion of critical conditions, refer to Time Scale in Section 3.1).

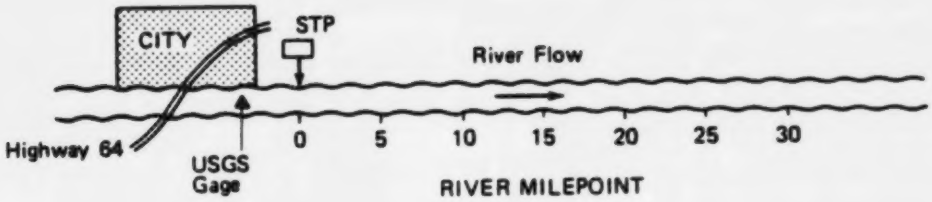
Critical conditions of dissolved oxygen and un-ionized ammonia concentration in the river occur during the summer when the flow is low and the river temperature is high. From eleven years of river temperature data collected as part of a limited river monitoring program by treatment plant personnel, the maximum average monthly river temperature is 27°C and occurs in August.

Therefore, design critical river conditions for water quality impact analyses are a river flow of 30 cfs and a river temperature of 27°C.

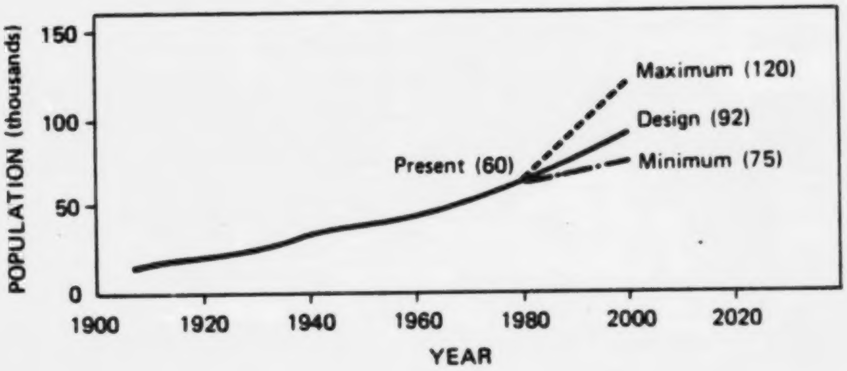
Note that for the example, both the critical low flow ($_7Q_{10}$) and the maximum average monthly temperature have been used in the projection, even though historical records (summarized in Figure 2-13) show minimum average monthly flow and temperature to occur in different months. This tacitly assumes that although the minimum average monthly flow occurs in September, the critical $_7Q_{10}$ could occur in August, the month of maximum average temperatures. In areas where it can be shown that the $_7Q_{10}$ will occur in a month with lower temperature, then the appropriate combination should be used rather than each of the extreme values. For example, critical low flows frequently occur during October in the northeast. An appropriate approach in such cases would be to define the $_7Q_{10}$ and temperature condition for each of the critical months (say June-October), determine which month is most critical, and use that month in WLA calculations.

For this example, assume that three surveys have been conducted that measured stream cross-sectional area under different flow regimes. Cross-sections were measured at 20 locations within the 30-mile long study area. From cross-sectional area measurements, it is concluded that the river is relatively uniform in the study area and, therefore, one average cross-sectional area and length can characterize the study area for each flow condition. If the river cross-sectional area varied significantly with distance, the river would have been divided into smaller reaches, each of which would have approximately uniform geometry for each prevailing flow condition. Dye study techniques provide an alternative means of accurately determining average velocity for a given river section.

A. STUDY AREA


CITY
STP
River Flow
Highway 64
USGS Gage
0
5
10
15
20
25
30
RIVER MILEPOINT

B. POPULATION PROJECTIONS



C. RIVER CLASSIFICATION AND USE

1. State Classification–B1
2. Designated Use–Fish and Wildlife Propogation
3. Water Quality Standards (partial)
    a. DO Concentration–Greater than 5.0 mg/ℓ
    b. Un-ionized Ammonia–Less than 0.02 mg/ℓ
4. Activities and Use
    a. Active and Locally Popular Fishery
    b. Several Campgrounds and State Park Have River as an Attraction
5. Problems
    No Documented Problems; Limited Water Quality Data Do Not Show Violations

Figure 2-11. Problem setting.

A. TREATMENT FACILITIES

Present: Trickling Filter Plant Constructed in 1958
Plant Now at Design Capacity of 7.5 MGD
Effluent Does Not Meet NPDES Permit for
Secondary Treatment

Proposed: Activated Sludge System to Provide
Secondary Treatment

B. EFFLUENT CHARACTERISTICS

		Present	Design①
Flow	MGD	7.5	11.5
BOD_5	mg/ℓ lb/day	40 2502	30 2877
$CBOD_u$②	mg/ℓ lb/day	80 5004	60 5754
NH_3-N③	mg/ℓ lb/day	15 938	15 1439
NBOD	mg/ℓ lb/day	68 4221	68 6475

① Preliminary Basis–Standard Secondary Treatment

Note that the 30 mg/l BOD_5 used in this example represents the secondary treatment effluent standard, and that better performance may occur during warm weather. Therefore, effluent characteristics used in modeling real-life situations should reflect the performance expected of the proposed facility during the critical period.

② Long-term BOD Tests Indicate Ratio of $CBOD_u/BOD_5 = 2.0$

③ NBOD = Stoichiometric Oxygen Requirements for Oxidation of Reduced Nitrogen Forms = 4.57 x NH_3-N (effluent oxidizable organic nitrogen is negligible)

Figure 2-12. Treatment facilities and effluent characteristics.

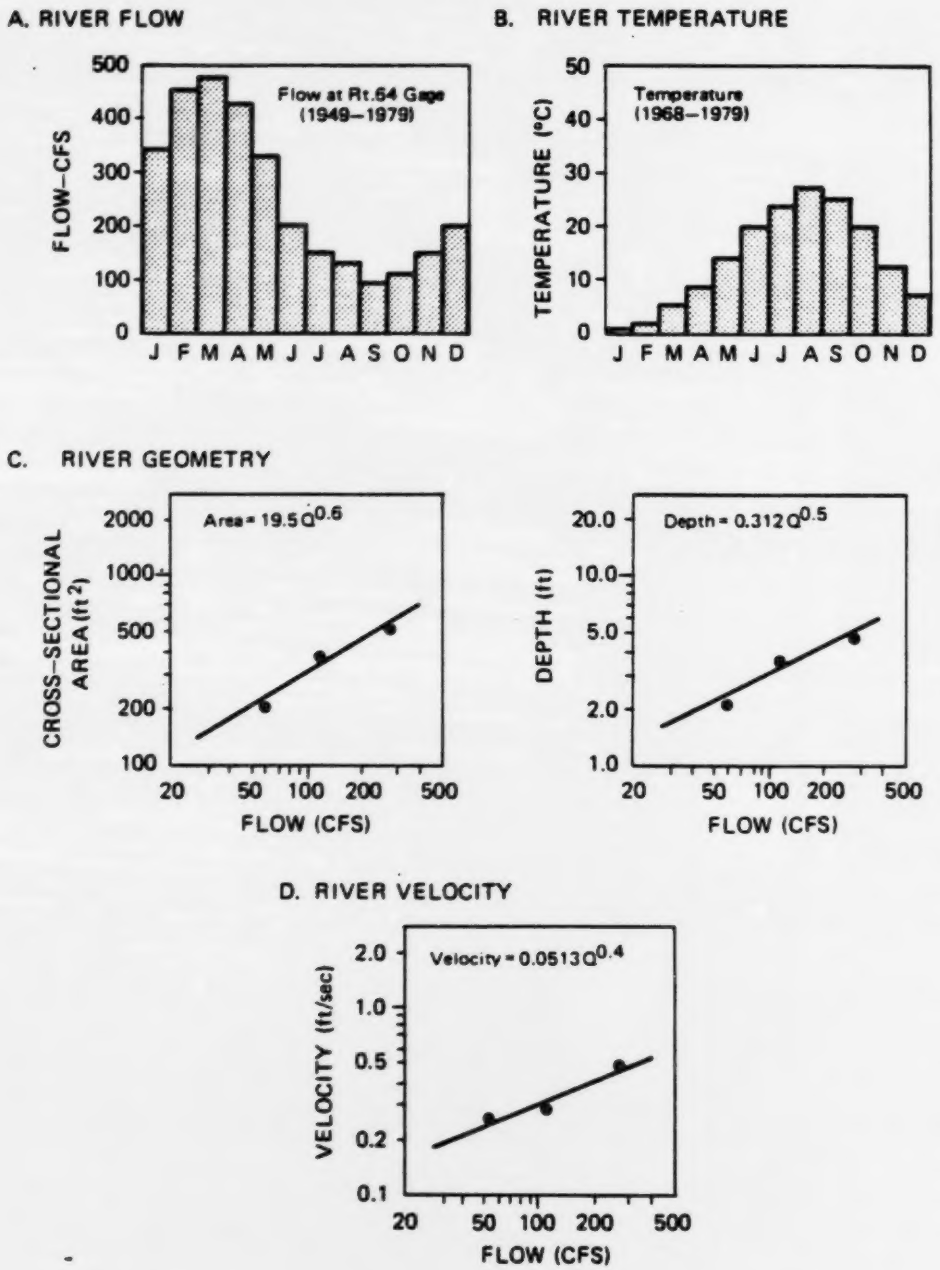


Figure 2-13. River flow, temperature, geometry, and velocity.

The average river velocity during each of the cross-sectional area survey periods was computed by applicaton of the continuity equation VELOCITY = FLOW/AREA. The average flow for each survey period was obtained from USGS records.

River cross-sectional area, depth and velocity generally form linear correlations with flow when the data are plotted on log-log scales. Figure 2-13 presents these log-log plots for the example problem. Interpolations and extrapolations of river geometry and velocity at specific flows can be made directly from the log-log plots or computed from the equation of the line of best fit. The equation for the line of best fit has the form: $Y = IQ^s$, where I is the intercept at Q = 1 cfs and s is the slope scaled directly from the plot (inches/inch). As shown in Figure 2-13, these log-log relationships are summarized as follows:

$$\text{AREA (ft}^2) = 19.5\ Q\ (\text{cfs})^{0.6} \qquad (2\text{-}21)$$

$$\text{DEPTH (ft)} = 0.312\ Q\ (\text{cfs})^{0.5} \qquad (2\text{-}22)$$

$$\text{VELOCITY (fps)} = 0.0513\ Q\ (\text{cfs})^{0.4} \qquad (2\text{-}23)$$

Using the above equations, river area, depth and velocity can be computed for any river flow. If river geometry data are available at only one flow regime, the relationship presented in Section 2.2 (equations 2-3, 2-4, and 2-5) would be used to calculate river depth, area, and velocity at other flows.

Review of River Water Quality Data. Historical river water quality data within the study area are limited. As part of the state environmental department's overall monitoring program for this river basin, water samples are periodically collected at stations located at river milepoints 11 and 25. These data represent approximately one grab sample per month during the summer over a five-year period. A review of these data does not reveal any water quality problems with regard to dissolved oxygen and un-ionized ammonia. Because there is no evidence of a water quality problem, only secondary treatment at the expanded plant has been proposed. No additional funds for Advanced Treatment (AT) have been proposed.

Considering the relatively good water quality, an appropriate level of effort for a waste load allocation (WLA) study initially can be limited to the analysis of a single river water quality data set collected during summer low flow conditions. Accordingly, a survey was conducted during two days in August when the river flow averaged 100 cfs and the river water temperature was 25°C. The results of this survey are presented in Figures 2-14 and 2-15.

The dissolved oxygen data in Figure 2-14, both August 1979 data and historical data, show river dissolved oxygen levels above the standard of 5.0 mg/l. The increase in river BOD_5 and ammonia concentrations at milepoint 1.0 show the impact of the treatment plant discharge. The gradually decreasing ammonia profile and increasing nitrite plus nitrate profile suggest that nitrification is occurring in the river. There is evidence that a natural denitrification process, in which nitrate and some oxygen-demanding material are removed from the water, may occur in some streams. At this time the natural process is neither fully understood nor proven to exist.

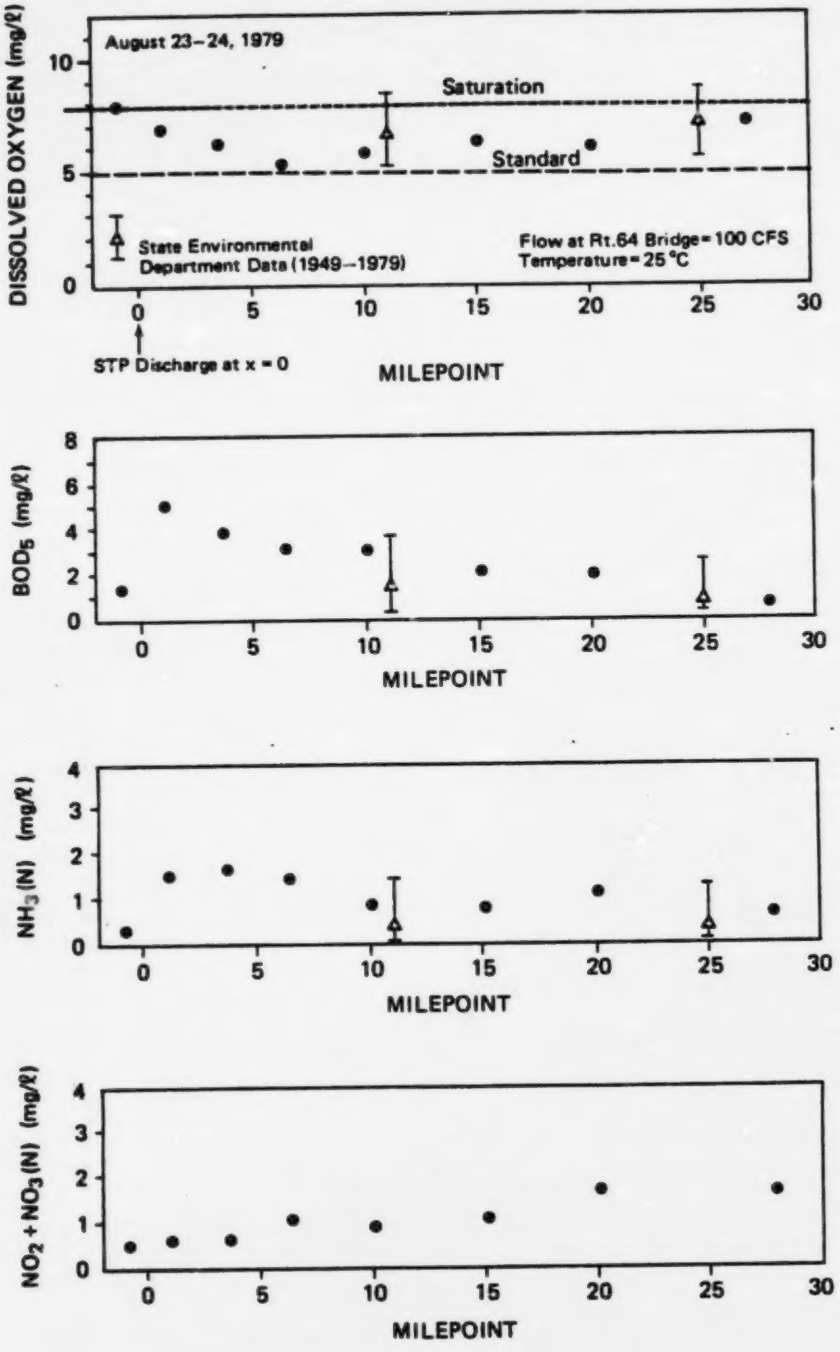


Figure 2-14. Dissolved oxygen, BOD, and nitrogen data (August 23-24, 1979).

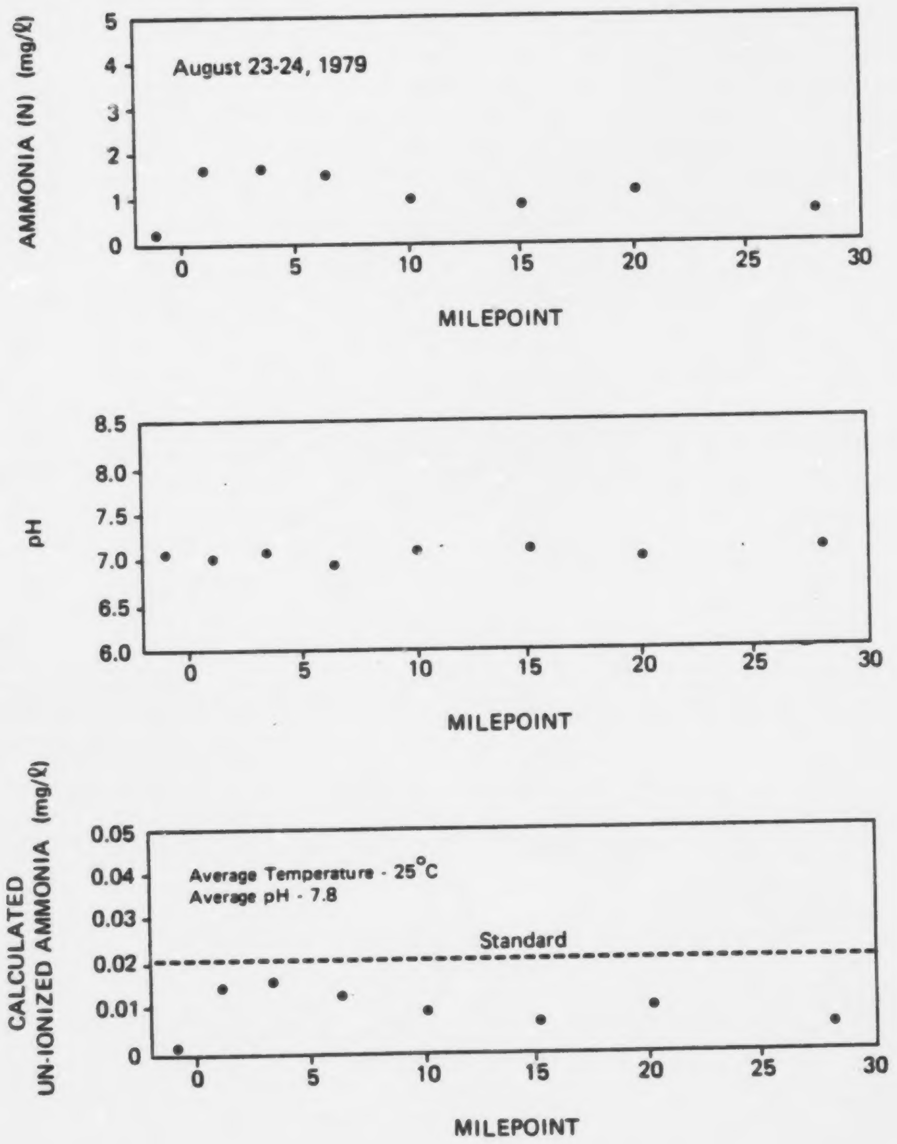


Figure 2-15. Ammonia, pH, and un-ionized ammonia data.

The concentration of un-ionized ammonia, the species of ammonia that is toxic to biological life, is dependent on the total river ammonia concentration, river pH, and river temperature. The higher the river pH and temperature, the higher the percentage of total ammonia that is in the un-ionized form. Figure 2-15 presents the measured total river ammonia concentration. The un-ionized ammonia concentration is determined from the relationship summarized in Figure 2-16, which relates un-ionized ammonia to pH and temperature. For a river temperature of 25°C and a pH of 7.2 (point A), un-ionized ammonia is 0.9 percent of the total ammonia concentration. During the August 1979 survey, the river un-ionized ammonia concentration was less than the standard of 0.02 mg/l.

Model Calibration Analysis. For this analysis, model calibration is the determination of the coefficients (reaction rates) of the equations (some of which were previously presented in sections 2.2 and 2.3) that describe the spatial distribution of BOD, ammonia, nitrite plus nitrate, and dissolved oxygen. The equations for each of these constituents are summarized as follows:

BOD_5

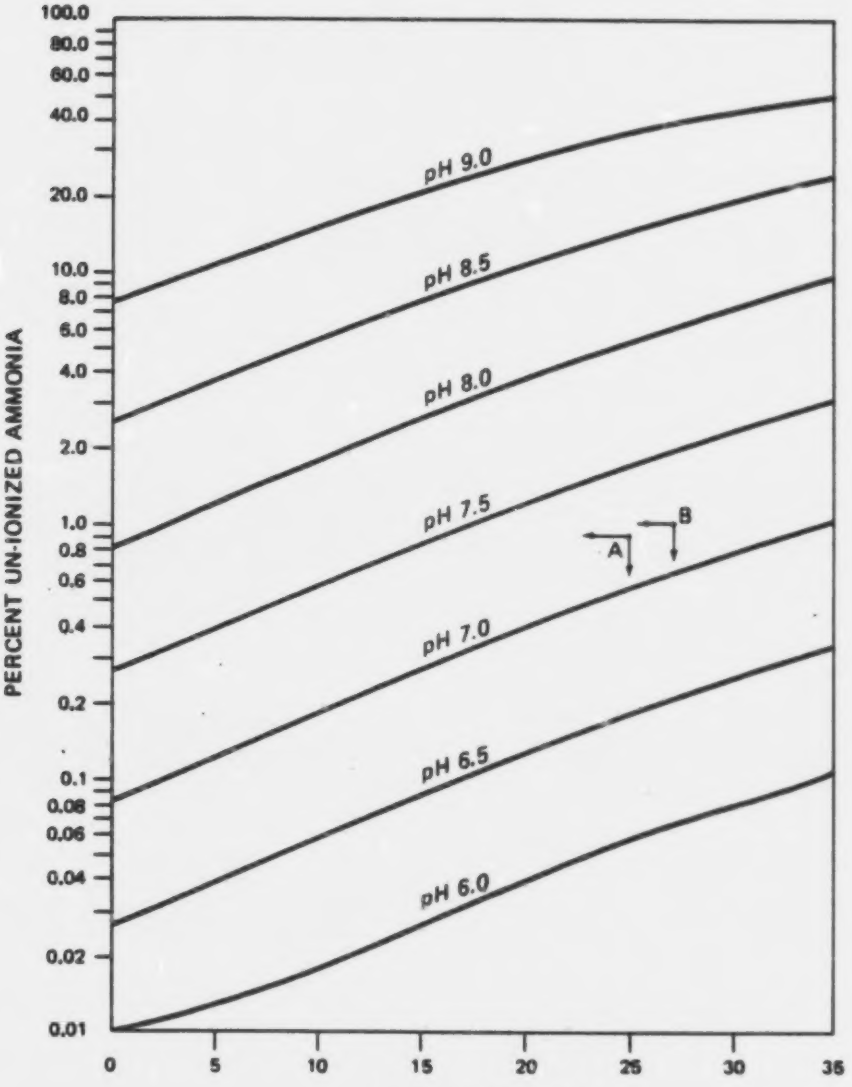
$$L_5 = (L_5)_o \, e^{\frac{-K_r}{U} x} \qquad (2\text{-}24)$$

Ammonia (as N)

$$NH_3 = (NH_3)_o \, e^{\frac{-K_n}{U} x} \qquad (2\text{-}25)$$

Nitrite plus Nitrate (as N) (2-26)

$$(NO_2 + NO_3) = (NO_2 + NO_3)_o + \left[(NH_3)\ (1 - e^{\frac{-K_n}{U} x}\right]$$



Note: Redrawn from
William T. Willingham
Ammonia Toxicity
USEPA 908/3-76-001
Feb. 1976 (4)

Figure 2-16. Percentage of unionized ammonia in ammonia-water solution at various pH and temperature values.

Dissolved Oxygen Deficit

$$D = \frac{K_d\ L_o}{K_a - K_r}\left[e^{\frac{-K_r}{U}x} - e^{\frac{-K_a}{U}x}\right] + \frac{K_n N_o}{K_a - K_n}\left[e^{\frac{-K_n}{U}x} - e^{\frac{-K_a}{U}x}\right] + D_o\, e^{\frac{-K_a}{U}x} \qquad (2\text{-}27)$$

In all these equations, the variables with the zero subscript are the concentrations at $x = 0$ (after integrating the upstream concentration and the treatment plant load). U is river velocity and K_r, K_d, K_n, and K_a are the BOD removal rate coefficient, BOD oxidation rate, nitrification rate, and atmospheric reaeration rate coefficients, respectively. L_o and N_o represent (respectively) the ultimate carbonaceous BOD and nitrogenous BOD at $x = 0$, and D_o is the initial oxygen deficit at $x = 0$.

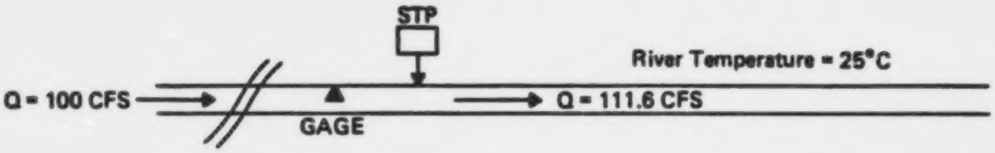
Note that the oxidation of ammonia (as expressed by equation 2-25) is analogous to the oxidation of BOD. Whereas the oxidation of carbonaceous BOD produces carbon dioxide as the end product, the end product of ammonia oxidation is nitrite plus nitrate. Thus, the equation for nitrite plus nitrate states that the nitrite plus nitrate in the river is equal to the concentration at $x = 0$ plus the amount gained from ammonia oxidation. When all forms of nitrogen are expressed as N, the terms in equation 2-26 are directly additive.

The coefficients in equations 2-24 through 2-27 are determined by the following steps. The river velocity is calculated for the August 23-24, 1979 river flow using equation 2-23. The initial conditions ($x = 0$) are determined from a mass balance with the upstream concentration and the plant load. The BOD removal coefficient (K_r) is the value of K_r that provides the best fit of the August 1979 BOD data with equation 2-24; for secondary effluents $K_d = K_r$ (no settling). The nitrification rate (K_n) is the value of K_n that simultaneously yields the best fit of both the ammonia and nitrite plus nitrate data with equations 2-25 and 2-26 respectively. The atmospheric reaeration rate is determined in accordance with Figure 2-6 in Section 2.2. In addition, long-term BOD tests have been performed and indicate the ratio of ultimate to 5-day carbonaceous BOD to be 2.0. A summary of these calculations is shown in Table 2-2, and the calibration results are presented in Figure 2-17.

In this example, the calculation of the dissolved oxygen profile agrees with the measured data quite favorably and without any adjustments. Often the calculated dissolved oxygen profile does not initially agree with the data due to other sources and sinks of oxygen, such as nonpoint source loads, algal photosynthesis and respiration, and benthal oxygen demands. For the sake of simplicity, these complications have been omitted. Section 3 includes a discussion of procedures to use when these complications are present.

Model Projections. Having calibrated a model for BOD, DO, and nitrogen, i.e., defining site-specific coefficients and accepting that some reservations on reliability exist because the model was not tested against an independent data set, the model may be used to project water quality impacts that might be expected under conditions of interest. Two specific cases of interest are the water quality impacts of existing wastewater loads and for future design wastewater loads for the minimum river flow (${}_7Q_{10}$) of 30 cfs. Dissolved oxygen, ammonia, and un-ionized ammonia profiles are presented for both cases in Figures 2-18 and 2-19. The model calculations for the design load condition are shown in Table 2-3.

TABLE 2-2. CALIBRATION ANALYSIS



UPSTREAM CONDITIONS

$BOD_5 = 1.0$ mg/ℓ
$NH_3(N) = 0.2$ mg/ℓ
DO = 8.2 mg/ℓ (SAT.)

PLANT EFFLUENT

Q = 7.5 MGD (11.6 CFS)
$BOD_5 = 40$ mg/ℓ (2502 lb/day)
$NH_3(N) = 15$ mg/ℓ (938 lb/day)
DO = 8.2 mg/ℓ (SAT.)

A. DETERMINATION OF RIVER DEPTH AND VELOCITY

(from equations in Figure 2-13)

Depth = $0.312\ Q^{0.5} = 0.312\ (111.6)^{0.5} = 3.3$ ft.
Velocity = $0.0513\ Q^{0.4} = 0.0513\ (111.6)^{0.4} = 0.34$ ft./sec. (5.6 mi./day)

B. DETERMINATION OF REACTION RATES

1. Reaeration (from Figure 2-6 or $K_a = 13\ U^{1/2}/H^{3/2}$)

$K_a = 13\ (0.34)^{1/2}/(3.3)^{3/2} = 1.26$/day at 20°C
K_a at 25°C = $1.26 \times 1.024^5 = 1.42$/day

2. BOD Removal and Oxidation Rates (from fit of river BOD data)*

$K_r = K_d = 0.30$/day at 20°C
K_r and K_d at 25°C = $0.30 \times 1.047^5 = 0.38$/day

3. Nitrification Rate (from fit of river ammonia and nitrate data)*

$K_n = 0.15$/day at 20°C
K_n at 25°C = $0.15 \times 1.08^5 = 0.22$/day

* Note: Since data were collected at river temperature of 25°C, K_d and K_n determined from data fit would be 0.38 and 0.22 respectively. It is assumed here that analyst has converted to rates at 20°C for convenience (a) in comparing with published data, and (b) in subsequent calculations which may be at temperatures other than 20°C and 25°C.

C. INITAL CONDITIONS (at x = 0)

$$\text{Concentration}_o = \frac{(\text{Upstream concentration (mg/l)} \times \text{Upstream flow (cfs)}) + (\text{Plant load (lb/day)} \times 0.1854)}{(\text{Upstream flow (cfs)} + \text{Plant flow (cfs)})}$$

(Note: 0.1854 is a conversion factor having units of $\frac{\text{day} - \text{ft}^3 - \text{mg}}{\text{lb} - \text{sec} - \text{l}}$)

$$BOD_5(x=0) = (L_5)_o = \frac{(1.0 \times 100) + (2502 \times 0.1854)}{111.6} = 5.1 \text{ mg/l}$$

$$\text{Ultimate CBOD } (L_o) = 2 \times BOD_5 = 2 \times 5.1 = 10.2 \text{ mg.l}$$

$$NH_3 - N\ (x=0) = \frac{(0.20 \times 100) + (938 \times 0.1854)}{111.6} = 1.74 \text{ mg/l}$$

$$NBOD\ (x = 0) = 4.5 \times NH_3 - N = 4.5 \times 1.8 = 8.1 \text{ mg/l}$$

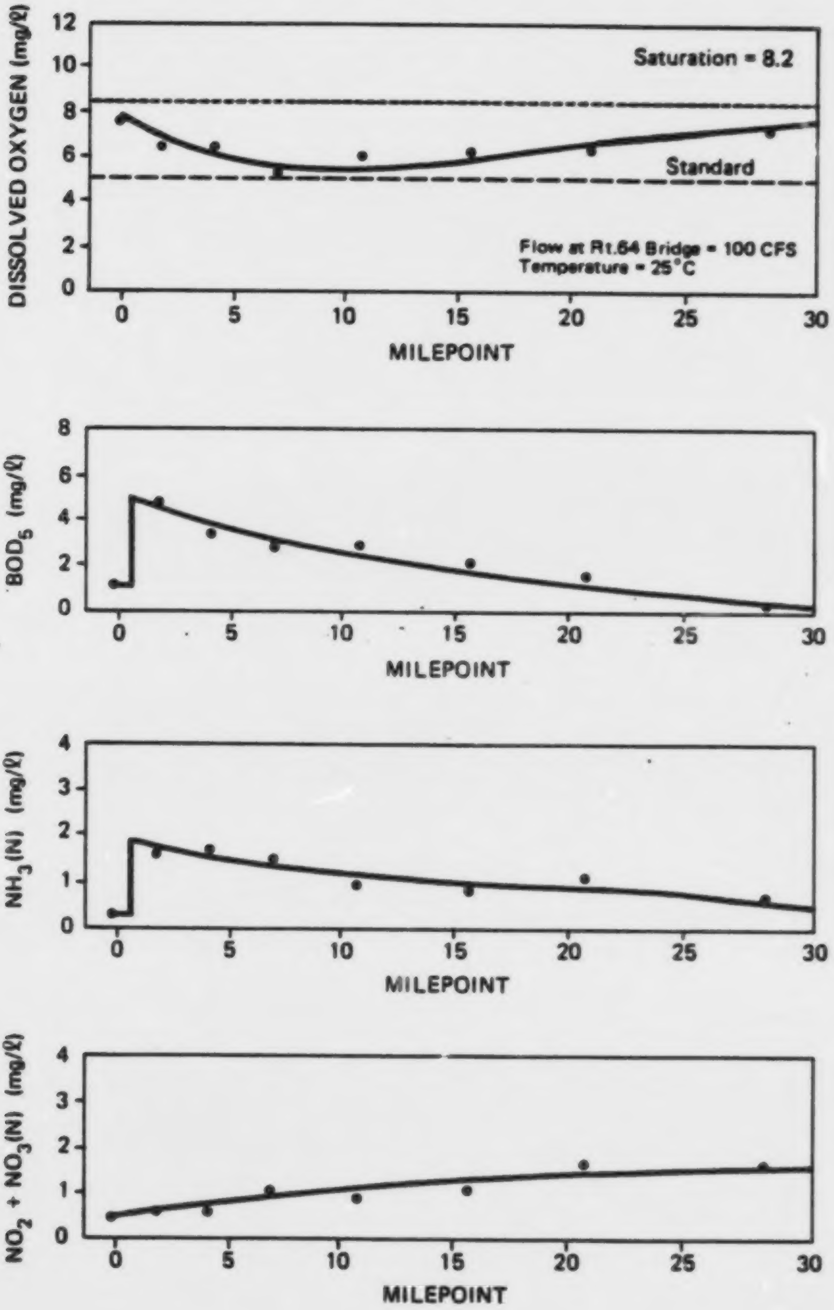


Figure 2-17. Model calibration analysis.

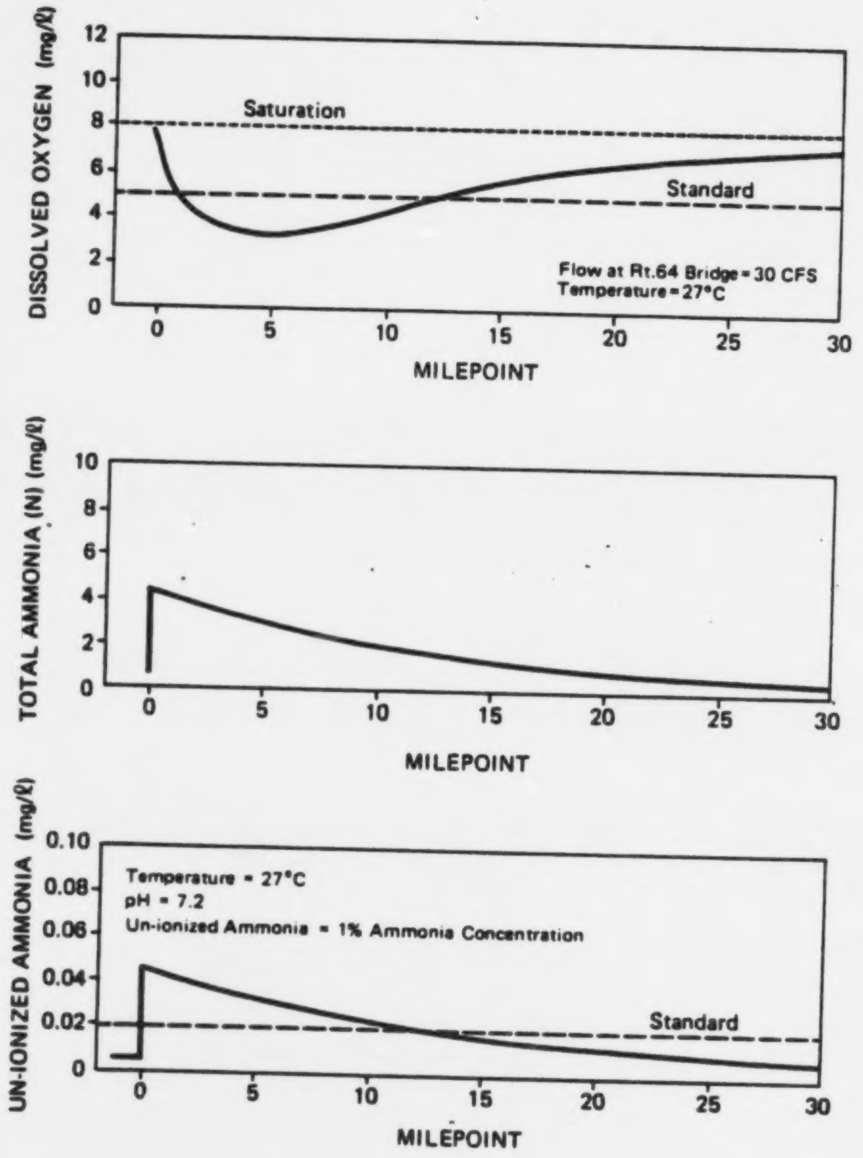


Figure 2-18. Projected dissolved oxygen, ammonia, and un-ionized ammonia (present wastewater load).

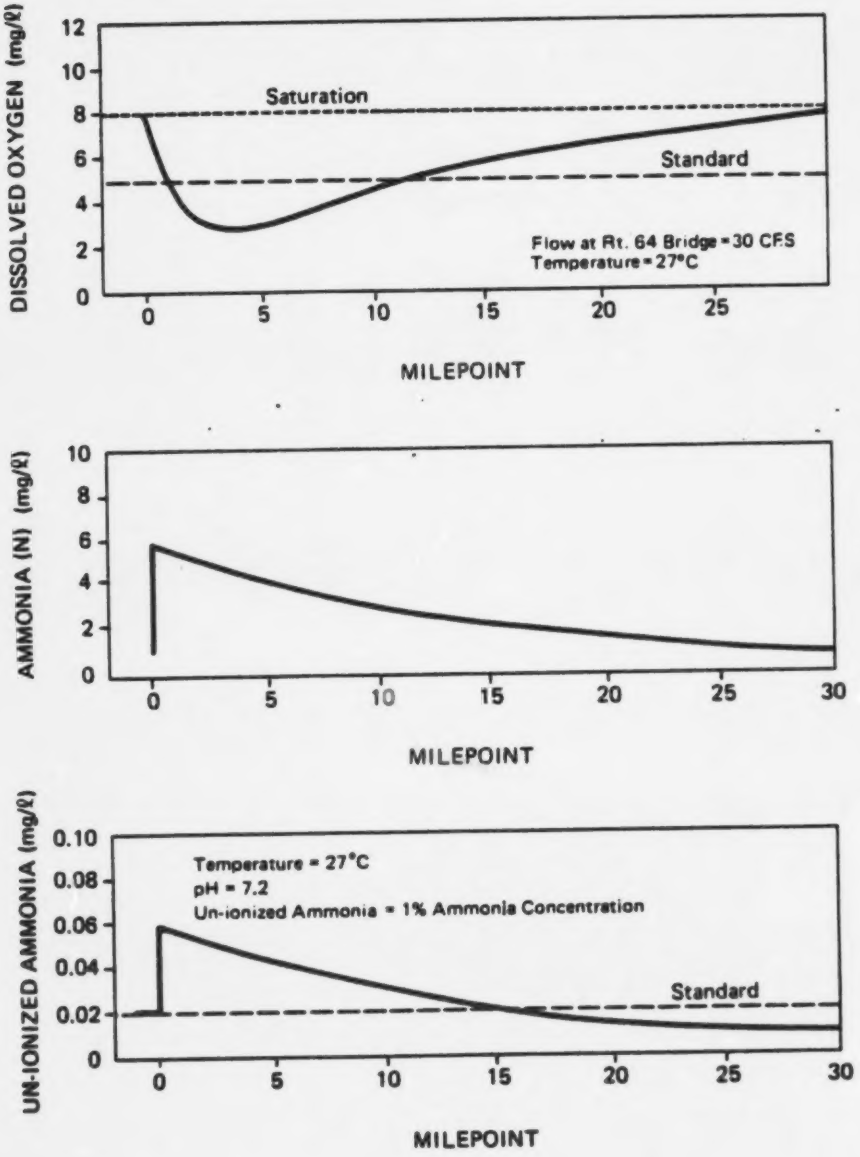
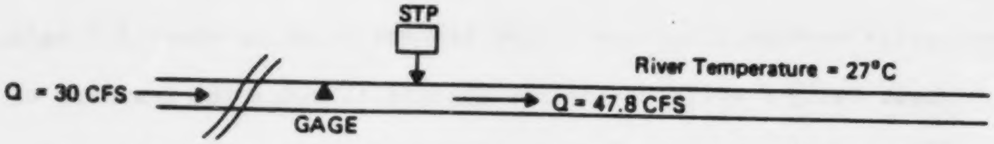


Figure 2-19. Projected dissolved oxygen, ammonia, and un-ionized ammonia (design wastewater load).

TABLE 2-3 PROJECTION ANALYSIS (design load)



UPSTREAM CONDITIONS

$BOD_5 = 1.0$ mg/ℓ
$NH_3(N) = 0.2$ mg/ℓ
DO = 8.0 mg/ℓ (SAT.)

PLANT EFFLUENT

Q = 11.5 MGD (17.8 CFS)
$BOD_5 = 30$ mg/ℓ (2877 lb/day)
$NH_3(N) = 15$ mg/ℓ (1439 lb/day)
DO = 8.0 mg/ℓ (SAT.)

A. DETERMINATION OF RIVER DEPTH AND VELOCITY
(from equations in Figure 2-13)

Depth = $0.312\ Q^{0.5} = 0.312\ (47.8)^{0.5} = 2.2$ ft.
Velocity = $0.0513\ Q^{0.4} = 0.0513\ (47.8)^{0.4} = 0.24$ ft./sec. (3.9 mi./day)

B. DETERMINATION OF REACTION RATES

1. Reaeration (from Figure 2-6 or $K_a = 13\ U^{1/2}/H^{3/2}$)

$K_a = 13\ (0.24)^{1/2}/(2.2)^{3/2} = 1.95$/day at 20°C
K_a at 27°C = $1.95 \times 1.024^7 = 2.30$/day

2. BOD Removal and Oxidation Rates (from Calibration Analysis)

$K_r = K_d = 0.30$/day at 20°C
K_r and K_d at 27°C = $0.30 \times 1.047^7 = 0.41$/day

3. Nitrification Rate (from Calibration Analysis)

$K_n = 0.15$/day at 20°C
K_n at 27°C = $0.15 \times 1.08^7 = 0.26$/day

C. INITAL CONDITIONS (at x = 0)

$$\text{Concentration}_o = \frac{(\text{Upstream concentration (mg/1)} \times \text{Upstream flow(cfs)}) + (\text{Plant load (lb/day)} \times 0.1854)}{(\text{Upstream flow (cfs)} + \text{Plant flow (cfs)})}$$

$$BOD_5\ (x=0) = \frac{(1.0 \times 30) + (2877 \times 0.1854) = 11.8}{47.8}$$

$$\text{Ultimate CBOD } (L_o) = 2 \times BOD_5 = 2 \times 11.8 = 23.6$$

$$NH_3 - N\ (x=0) = \frac{(0.20 \times 30) + (1439 \times 0.1854)}{47.8} = 5.7$$

$$NBOD\ (x=0) = 4.5 \times NH_3 - N = 4.5 \times 5.7 = 25.7$$

The calculated profiles in Figure 2-18 show that present wastewater loads would result in dissolved oxygen and un-ionized ammonia water quality standards violations over approximately ten miles of river, if design drought flow conditions were to occur ($_7Q_{10}$ flow and a river temperature of 27°C). The lowest daily average dissolved oxygen concentration is about 3.0 mg/l, and the highest daily average un-ionized ammonia is twice the standard of 0.02 mg/l. At 27°C and with a pH of 7.2, un-ionized ammonia is 1 percent of the total ammonia concentration (point B - Figure 2-16).

Calculated dissolved oxygen and un-ionized ammonia profiles for $_7Q_{10}$ flow conditions and the design wastewater load are presented in Figure 2-19. For the design load, the carbonaceous BOD is only slightly greater than the present load, but the ammonia and nitrogenous BOD loads are about 50% greater than present loads (see Figure 2-12). The minimum dissolved oxygen is about 2.6 mg/l, and the maximum un-ionized ammonia about 0.06 mg/l.

The projected dissolved oxygen and un-ionized ammonia profiles in Figures 2-18 and 2-19 indicate that whether or not the treatment plant expands from 7.5 MGD to 11.5 MGD, some reduction of wastewater BOD (carbonaceous and/or nitrogenous) and ammonia is required to meet water quality standards during critical low flow conditions. One method of computing the required reduction in wastewater BOD and ammonia is to calculate a series of dissolved oxygen and un-ionized ammonia profiles and, through trial and error, arrive at the proper combination of wastewater load reductions that meets water quality standards. An alternative to the trial and error method for dissolved oxygen is to separately calculate the dissolved oxygen deficit due to each BOD source (upstream BOD, plant carbonaceous BOD, plant nitrogenous

BOD). This has been done for the design wastewater load case and is shown in Figure 2-20. (See also section 3.5 for further discussion and examples.)

The top profile is the calculated dissolved oxygen distribution in which the lowest daily average dissolved oxygen concentration of 2.6 mg/l (maximum deficit of 5.4 mg/l) occurs at milepoint 4. The next three profiles are the components of the total dissolved oxygen deficit produced individually by the oxidation of upstream BOD, treatment plant carbonaceous BOD, and treatment plant nitrogenous BOD, respectively. On each deficit profile, the deficit produced at the critical point in the river, milepoint 4, is indicated. Note that the deficits of the component parts at milepoint 4 add up to the total deficit of 5.4 mg/l. Inspection of equation 2-27 shows that the carbonaceous and nitrogenous deficits are additive. The upstream BOD may be considered as a fraction of the total carbonaceous BOD in the river and thus separable from the plant carbonaceous BOD.

Knowing the relative contribution of each BOD source to the total deficit, it is an easy task to select combinations of wastewater BOD reductions that will achieve water quality standards. For the sake of simplicity, the use of a safety factor in the allocation procedure is omitted in this example. However, the first example in Section 3.5 demonstrates the application of a margin of safety. At 27°C, dissolved oxygen saturation is 8.0 mg/l; therefore, for a dissolved oxygen standard of 5.0 mg/l, the allowable maximum deficit is 3.0 mg/l. Assuming that the upstream BOD deficit of 0.2 mg/l is uncontrollable, 2.8 mg/l of deficit would be available for the total of carbonaceous and nitrogenous plant BOD oxidation. Considering the un-ionized ammonia standard and the economics of nitrification versus advanced carbonaceous BOD removal suggests that

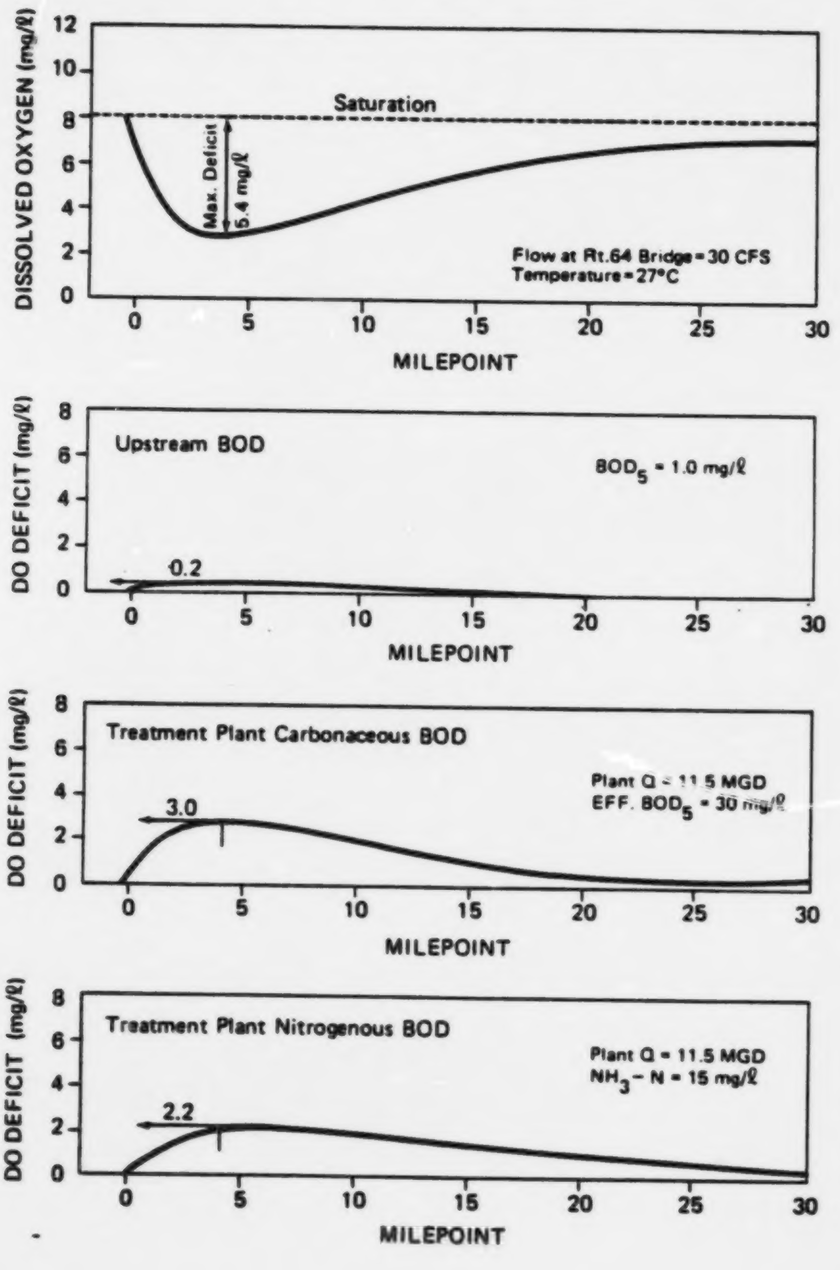


Figure 2-20. Dissolved oxygen component unit responses.

providing nitrification facilities for the expanded treatment plant is a cost-effective step towards achieving water quality standards. Assuming that nitrification removes 90% of the ammonia (i.e., nitrogenous BOD) the un-ionized ammonia standard would be met and the nitrogenous BOD deficit would be reduced to 0.2 mg/l, yielding a total deficit of 3.4 mg/l (DO of 4.6 mg/l) at milepoint 4.

Although calculations show that nitrification and standard secondary treatment for the design plant flow of 11.5 MGD will not meet dissolved oxygen standards, the waste load allocation analysis should not be carried any further with currently available information. Planning for a summer nitrification facility can proceed since the process provides for a significant reduction in ultimate BOD at a relatively low cost compared to other types of AT. However, before the model is used to calculate the additional carbonaceous BOD removal beyond secondary treatment that may be required, some additional steps should be taken. First, the model should be calibrated and validated against one or two more data sets to check the model coefficients under different flow regimes, especially lower flows than the first survey, if possible. A sensitivity analysis should be performed to relate the cost of required wastewater treatment to changes in population estimates. For example, if the population in the year 2000 is 75,000 people (projected minimum), will additional carbonaceous BOD treatment be required? A sensitivity analysis of the effect of different reserve policies on required treatment should also be performed. For example, what is the increase in wastewater treatment costs if the dissolved oxygen reserved for future development is set at 0.5 mg/l versus 0.25 mg/l? It is clear from these questions that a final waste load allocation should be the result of more than a model projection. Such a decision takes many factors into consideration, one of which is the impact of a design wastewater load on water quality.

SECTION 3.0

MODELS: SELECTION AND USE

3.1 SELECTING A MODEL

Introduction

Three basic components collectively constitute a site-specific dissolved oxygen water quality model. These are:

- The types of reactions which are included (i.e., the phenomena considered) and the equations used to represent these reactions. Examples of phenomena that may be included in a water quality model include: oxidation of BOD in the water column, oxygen demand from benthal deposits, oxidation of ammonia, atmospheric reaeration, diurnal changes in oxygen production and depletion (P-R) due to algal activity, provision for input of distributed loads, etc.
- The calculation framework. This may range from a single equation providing an analytical solution for the DO deficit to the array of equations, calculation instructions and input/output processes that collectively comprise the software of a generalized receiving water quality model (DOSAG, QUAL II, etc.).
- The site-specific values assigned to the rate coefficients employed in the calculations, stream geometry, the location of input loads, etc., i.e., the details that convert the generalized model framework to a model of a specific receiving water system.

All three components are necessary to define a site-specific water quality model.

The preferred approach is to use the simplest model that can be applied in a particular case. Ideally, the model should include only those phenomena that are operative and important in the river or stream being modeled. The most appropriate procedure for selecting a model is to first define the phenomena that are important for the particular site-specific analysis to be performed. Activities that help to define phenomena that should be incorporated include the following: (a) reviews of existing data on waste loads, stream flows, and water quality; (b) preliminary mass balance calculations using simple models or equations that provide analytical solutions for various load sources (combined sewer overflows, nonpoint sources, sediment) and reaction phenomena (nitrification, algal photosynthesis and respiration, etc.).

It is also desirable to attempt to anticipate the technical issues with respect to control actions (level of treatment, alternate discharge locations, etc.) and determine whether this will influence the types of reactions that will be important. From the foregoing, the analyst will generally be able to establish the phenomena that should be included in the selected model and the time and space scale of the analysis which is most appropriate.

Under ideal circumstances, one would select a formal model or analysis approach that included all the phenomena determined to be important in the study area, and which excluded those reactions that are insiginificant in the case in question. Qhile this guidance should be followed as much as possible, in practice a calculation framework or model often will be selected because it is available or familiar to the analyst.

In such cases, two criteria are important to apply. First, the model selected must be capable of handling all of the important site-specific phenomena considering the time and space scale of the analysis and using the equations and formulations specified. Secondly, provision should be made, where possible, to eliminate from the calculation framework the effect of any phenomenon that are insignificant in the site-specific analysis. In some cases inclusion of phenomena judged to be unimportant on a site-specific basis can increase the level of uncertainty of the analysis and thus directly affect decisions. In these situations, additional data collection, sensitivity runs, and other aspects of the overall waste load allocation program must be considered, in order that phenomena contained in the calculations are adequately addressed.

Additional evaluation criteria for model selection include completeness of computer program documentation, costs for manpower, and computer time.

The third component of the water quality model is the most significant in the context of decision making. The activities of model calibration and verification, which are discussed in Section 3.3, are key elements that essentially consist of comparisons of observed water quality and calculated responses under a range of conditions that are diverse enough to test the level of understanding of individual phenomenon and define site-specific model coefficients and parameters. In this regard, data collection programs form an essential element of the analysis.

The overall process of modeling is discussed in the context of the three components of a water quality model. First, some general and specific considerations that should be addressed in selecting a model are discussed. Next, available models and their important features are described, and finally an approach and considerations for calibrating and verifying a model are presented.

Important Considerations in Selecting a Model

Selecting the Number of Dimensions. Most waste load allocation projects that address dissolved oxygen in streams and rivers employ one-dimensional steady-state calculations. Both theory and experience demonstrate that dissolved oxygen gradients in streams and rivers are largest along the longitudinal axis of the system with relatively minor variations in the lateral and vertical dimensions. This results from the time scale for the dissolved oxygen response. Generally, near-field analysis is not important and only far-field problems need to be examined (i.e., mixing zone calculations are usually ignored). The number and definition of spatial dimensions that must be considered can usually be determined by an examination of historical water quality data. Field data to be examined should include dissolved oxygen and other variables such as temperature, conductivity, BOD, NH_3, etc.

Certain river situations may require a framework that encompasses a two-dimensional analysis. These situations are generally associated with deep rivers or run of the river impoundments when vertical or lateral gradients can be significant. Depending on the geomorphology, the upstream regions of lakes and impoundments may be characterized by significant lateral, as well as longitudinal, variations in dissolved oxygen that would require a two-dimensional analysis.

Water quality observations and the problem setting will determine the number of spatial dimensions required for a site-specific analysis. In particular, if a second dimension is considered, the investigator should provide justification in terms of the specific decision-making elements with regard to controls and treatment that require the information resulting from inclusion of a second dimension in the analysis. This requirement is necessary since the additional dimension in the analysis for streams or rivers will usually require substantially more data collection efforts and generally will result in generation of "free" modeling parameters whose site-specific values cannot be determined reliably. Thus, the additional dimension can tend to weaken the analysis and may adversely affect the ability to make decisions.

Three-dimensional analysis of stream and river systems is considered to be more of a research and development activity and generally does not appear appropriate for inclusion in the context of projects addressing WLA decisions.

Loads, Sources, and Sinks. Loads, sources and sinks that can influence the dissolved oxygen distribution in streams and rivers are:

- point source discharges from waste treatment plants
- urban runoff from combined and separate sewer systems
- nonpoint sources
- sediment oxygen demand
- oxygen production and utilization by phytoplankton or aquatic weeds
- upstream sources of oxygen demand or dissolved oxygen deficit

These will be collectively defined as sources in this subsection.

All sources that are explicitly included in the analysis require direct measurements on appropriate time and space scales to define the magnitude of the individual source by contaminant. In addition, as discussed in Section 3.2, receiving-water quality data is required under diverse conditions in order to evaluate the effect of individual sources. Therefore, it is suggested that each project create two categories for sources. Category I should consist of major sources controlling water quality under existing and projected conditions, and all sources that are to be controlled. Sources in this category will require thorough data collection and modeling analysis to define both the magnitude and effect of the individual sources. Category II will contain background sources; small to moderate data collection and analysis would be required for sources in this category. The basic distinction is that sources in the second category should not influence decisions on load allocations; i.e., even if the magnitude and effect of individual sources in category II were incorrectly identified, the basic decisions resulting from the project would not change. This requires that category II sources be a small portion of the total. Historical data on sources can be combined with information in the literature and calculations to place sources in the appropriate category.

The primary contaminants of concern associated with sources are organic carbon compounds that produce carbonaceous biochemical oxygen demands (CBOD) and the reduced forms of nitrogen that produce nitrogenous biochemical oxygen demands (NBOD). For each source type, it is necessary to define the magnitude of the ultimate oxygen demand for both classes of contaminants. Long-term BOD data are required. The carbonaceous demand alone is measured by using a nitrification-inhibited test. In addition, experimental information may be required to distinguish between the forms of organic nitrogen that can hydrolize to ammonia and the nitrogen that is, in a sense, refractory and is not transformed to ammonia. This distinction can be important if nitrification is of concern. The effluents from treatment plants without nitrification can contain potentially significant concentrations of organic nitrogen. The degree of nitrification required can be influenced by the organic nitrogen level in the effluent that can be transformed to ammonia and subsequently oxidized in the stream or river.

Time Scale. The time scale selected for the analysis should be a function of both the observed water quality and the dissolved oxygen standards or criteria for the system being analyzed. Dissolved oxygen analysis in streams and rivers usually can be performed on a seasonal time scale, employing either steady-state or time-variable analysis. It is desirable to evaluate water quality data collected during several seasons to determine the critical period to be analyzed. The most frequent critical period is the low-flow, high-temperature summer period. However, winter periods may also be critical when ice cover is important. In some situations, the fall may be significant if upstream sources of organic carbon from phytoplankton and/or aquatic weeds result in large depressions in dissolved oxygen levels. Also, spring floods that pick up large amounts of organic debris from adjacent floodplains may result in severe DO depletion.

Next, the analyst must determine the time interval to be used in the water quality analysis. Several choices available are listed below in order of increasing complexity:

- steady-state
- quasi steady-state
    - constant loads--constant stream flow--diurnal dissolved oxygen production by phytoplankton or aquatic plants
    - constant loads--variable stream flow
    - variable loads--constant stream flow
    - other combinations of the above
- fully time-variable analysis

In a steady-state analysis, a spatial profile of concentration is calculated, such as would result at equilibrium under a constant set of input conditions (stream flows, waste loads, temperature, etc.). To the extent that actual variations in waste load, stream flow, etc. can be realistically approximated by constant conditions for the period covered by the analysis, the calculated receiving water concentration profile will approximate an average of the actual concentrations during that period.

A fully time-variable analysis performs successive calculations at relatively short time steps and accepts variable input values for parameters such as stream flow, waste load, and temperature. Its output is a record of both temporal and spatial fluctuations in the calculated water quality concentrations. Practical considerations of cost and operating time usually limit the duration that can be covered by such an analysis.

"Continuous" versions of time-variable models extend the calculations over longer periods of time by using larger time steps and averaging the variable input over that period. As a result, the calculated receiving water concentrations will not reflect short-term variations but should track the longer-term fluctuations.

Quasi steady-state analyses usually have one time-variable element incorporated in what remains as basically a steady-state calculation.

Table 3-1 presents a series of steps, related questions, and output that can be employed to determine the appropriate time interval for the analysis. The decision on the time interval to be used in the analysis is critical to overall project success and should be documented in terms of the impact on water quality control decisions. Complex analysis frameworks associated with time-variable (dynamic) models will require more data collection and increased study costs for model runs, input and output analysis, and sensitivity runs. The degree of uncertainty in the analysis can also increase due to "free"

Table 3-1. STEPS FOR DOCUMENTING INCLUSION OF TIME-VARIABLE OR QUASI STEADY-STATE WATER QUALITY ANALYSIS

Steps	Typical Questions	Output Documentation
Review existing water quality data Review flow data Review rainfall data.	When are standards violations observed? Are violations associated with diurnal fluctuations, rainfall, flow variation, or season of the year?	Plots of data Regression analysis or plots
Define probable or possible causes of observed quality problems. (Use experience supplemented by calculations.)	Do point source waste loads control water quality? What loading types control quality (event, continuous, etc.)? How important are factors other than loads? Is water quality controllable?	Tabulate loads and probable range of effects defined by calculations. Direct statement showing how control is possible
List typical control options. For control options, list information needed from modeling study to make decisions.	What planning or treatment decisions are affected by output from non-steady state analysis?	Tabulate decisions and information that requires non-steady state analysis

model parameters. In general, the least complicated analysis that will provide the information required to make water quality control decisions should be selected. As more complex analyses are built into a study, the requirements for documentation and justification in terms of decisions will increase.

Several levels of analysis are available for situations that are influenced by oxygen production and utilization from phytoplankton and/or aquatic weeds. Steady-state analysis may be employed using average oxygen production and respiration (P-R) terms. Quasi steady-state analyses are also available for streams and rivers (5) that use steady-state calculations supplemented by time-variable calculations of diurnal oxygen variations. Complex kinetic systems are also available that relate oxygen levels to phytoplankton populations (chlorophyll) that in turn are controlled by light, nutrients, zooplankton, etc. These latter frameworks are time variable and require extensive data for model calibration and verification. Combinations of modeling frameworks and data collection programs can provide a spectrum of analyis to fit most problem settings. In most situations, it is suggested that a steady-state or quasi steady-state analysis be considered as a reasonable framework. Because of the complexity of the technical and management issues involved in using time variable models, they should be used only in cases where sufficient data exists and where this level of understanding of water quality problems is essential to determining treatment needs.

The issue of the time interval of the analysis is in part controlled by the sources considered in category I. Point sources, sediment oxygen demand, and upstream conditions usually can be adequately represented by steady-state modeling, which employs time-averaged values for the loads from these sources.

The same type of analysis can be appropriate for some nonpoint sources, such as those associated with groundwater inflow, leaching from bottom deposits, and drainage not directly related to transient events such as storm runoff or spills. By contrast, event-related inputs of mass, such as those associated with storms that produce urban runoff and runoff from other land use types, can require either a time-variable analysis or a quasi steady-state analysis. The quasi steady-state analysis often can be considered in situations when the receiving water is large and the incremental flow associated with the study area being modeled is small. However, for most of these situations, a time-variable analysis has been necessary. While expensive, the time-variable analysis can be applied satisfactorily to analyze observed data provided sufficient data exists or can be obtained. Projections present a special set of problems in terms of identifying the storms or storm sequences to be used to develop the waste load allocation. Furthermore, the event-related dissolved oxygen problem can be influenced strongly by the hydrograph after the event and the geomorphology of the downstream segments of the water body. In addition, the basic technical, economic, and environmental issues associated with wet weather standards for dissolved oxygen have not yet been addressed fully.

As a result of these complexities, and in order to permit timely action on decisions that must be made regarding control of point source loads, EPA has adopted the following basic strategy for developing waste load allocations for point sources. As a first approach for the general case, WLA calculations will ignore storm-induced loads and the issue of wet weather standards and will concentrate on point source impacts under critical low-flow conditions. While it is recognized that wet weather effects and the possible need for control of Storm runoff loads need to be considered (at least in some locations, possibly in most) such considerations can be deferred until the results of current EPA investigations of these issues can be incorporated into policy guidance to be issued in the future.

This approach is most appropriate for the issue of BOD/DO effects in rivers and streams. In advective river systems (unlike estuaries) the residual effects of storm loads on BOD/DO can be ignored. Point source impacts under critical flow conditions, therefore, represent a design situation that must be addressed independently of any separate consideration that may ulitmately prove appropriate to apply to storm loads.

Spatial Extent. The spatial extent of the modeling analysis should be well into the zone of dissolved oxygen recovery. This spatial coverage should be considered for several reasons.

- Reaeration is a dominant factor in this region, and analysis can provide information on the value of the reaeration coefficient.
- In many situations, a key issue in decision making concerns the presence of nitrification and the rate at which it may occur following upgrading of treatment. Observations of nitrification in the zone of dissolved oxygen recovery could be valuable in defining bounds for nitrification rates to be considered for making projections under future conditions.
- Indications of the growth of phytoplankton and aquatic weeds, after treatment has been upgraded, often can be obtained by examining the dissolved oxygen recovery zone.

The information obtained from the analysis of the zone of dissolved oxygen recovery will depend, to a large extent, on the uniformity of system geomorphology.

Transport Systems. Dispersion is present, to some extent, in all bodies of water. However, water quality profiles, such as dissolved oxygen concentrations, may not be influenced when the dispersive mixing is small and/or the advective transport is large. In these situations, which will characterize most WLA projects on streams and rivers, decisions will not be influenced by inclusion of dispersion in the analysis. This will reduce the complexity of the calculations and data collection programs. The importance of dispersion is site-specific, and can be estimated by the following procedure:

STEP 1 - Calculate the approximate Longitudinal Dispersion Coefficient (Ref 6).

$$E = 0.011\, U^2 W^2 / H U_* \qquad (3\text{-}1)$$

where:

E = Longitudinal Dispersion Coefficient (ft/sec)
U = Average Stream Velocity (ft/sec)
W = Stream Width (ft)
H = Stream Depth (ft)
U_* = Shear Velocity (ft/sec)

The Shear Velocity (U^*) for many streams is approximately one tenth of the average stream velocity, and can be estimated by:

$$U_* = \sqrt{gHS}$$

where:

g = Gravitation Constant (32.2 ft/sec^2)
S = Stream Slope (ft/ft)

STEP 2 - Calculate the "estuary number (n)" as defined by O'Connor (7).

The longitudinal dispersion coefficient can be employed with stream velocity and reaction rate (K_d), to develop this dimensionless number.

$$n = \frac{K_d E}{U^2} \qquad (3\text{-}2)$$

The O'Connor Number (n), and the ratio ($\emptyset$) of the reaeration rate coefficient (K_a) to deoxygenation rate coefficient (K_d) . . .

$$\emptyset = K_a/K_d \tag{3-3}$$

can be used with Figure 3-0, to provide a basis for judging the significance of dispersion in calculations of dissolved oxygen concentration.

Inspection of this figure indicates that for advective streams with values for n of about 0.1 or less, neglecting dispersion effects will affect the calculation of the maximum dissolved oxygen deficit (critical deficit, D_c) by less than 10%. For WLA studies, dispersion can be ignored in such cases. Where reaeration is high relative to deoxygenation rates (high values of $\emptyset$), the lack of sensitivity to dispersion extends to higher values of n, as indicated by the essentially horizontal lines for the higher values of $\emptyset$.

It should be noted that the estimates of the dispersion coefficient, and the ratio of the maximum DO Deficit to the initial BOD concentration (D_c/L_o), incorporate several simplifying assumptions. The foregoing approach must therefore be considered to be an approximation. It should, however, be adequate for use in most WLA studies.

There may be situations where dispersion is considered significant by the investigator even though the foregoing analysis suggests otherwise. Examples could include swamps, tidal rivers, or upstream segments of impoundments. Detailed documentation defining the magnitude of dispersion and relating these phenomena to waste load allocation decisions should be developed early in the study to support inclusion of dispersion in the modeling studies. If the

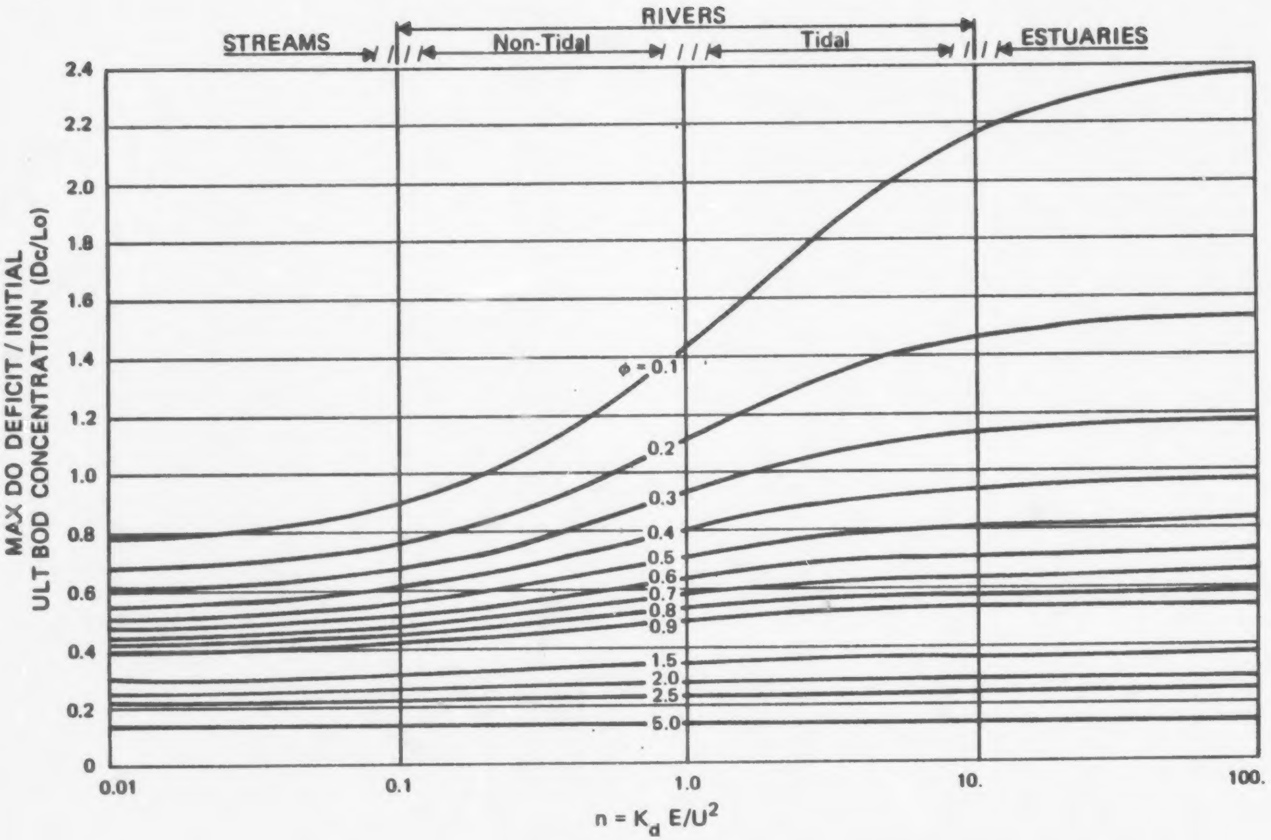


Figure 3-0. Dissolved oxygen response as a function of $K_d E/U^2$.

computational framework employed in the analysis introduces dispersion due to spatial segmentation or numerical approximations (called numerical dispersion or numerical mixing), the study should contain an evaluation of the influence of dispersion on calculations of water quality. Finally, the influence of dispersion on decisions and waste load allocations should also be supplied in this situation. The requirement relating to numerical mixing can often be met by comparisons of analytical solutions with computer output under comparable conditions.

A flow balance is required for the modeling effort, and, therefore, consideration should be given to the potential importance of groundwater inflow and outflow. In addition, flow from significant tributaries and waste sources must be included in the model. Each of the sources of flow included in the model must also be supported by data on the concentrations of significant constituents, such as dissolved oxygen, BOD, NH_3, etc.

Data on the cross-sectional area, depth and time of travel (or velocity), as a function of flow, are required for the flows at which observed water quality data are collected and at the critical flow regimes used for projections.

Variables to be Considered and Formulations. Several combinations of variables can be significant in dissolved oxygen analysis in streams and rivers. The most common are shown in Figure 3-1. Each of the four combinations of variables has been employed to develop waste load allocations. The selection should be based on site- and problem-specific factors. Documentation of the rationale for selection of a particular combination of variables should be provided in an early stage of the study and should include an examination of observed water quality data, considering each variable supplemented by

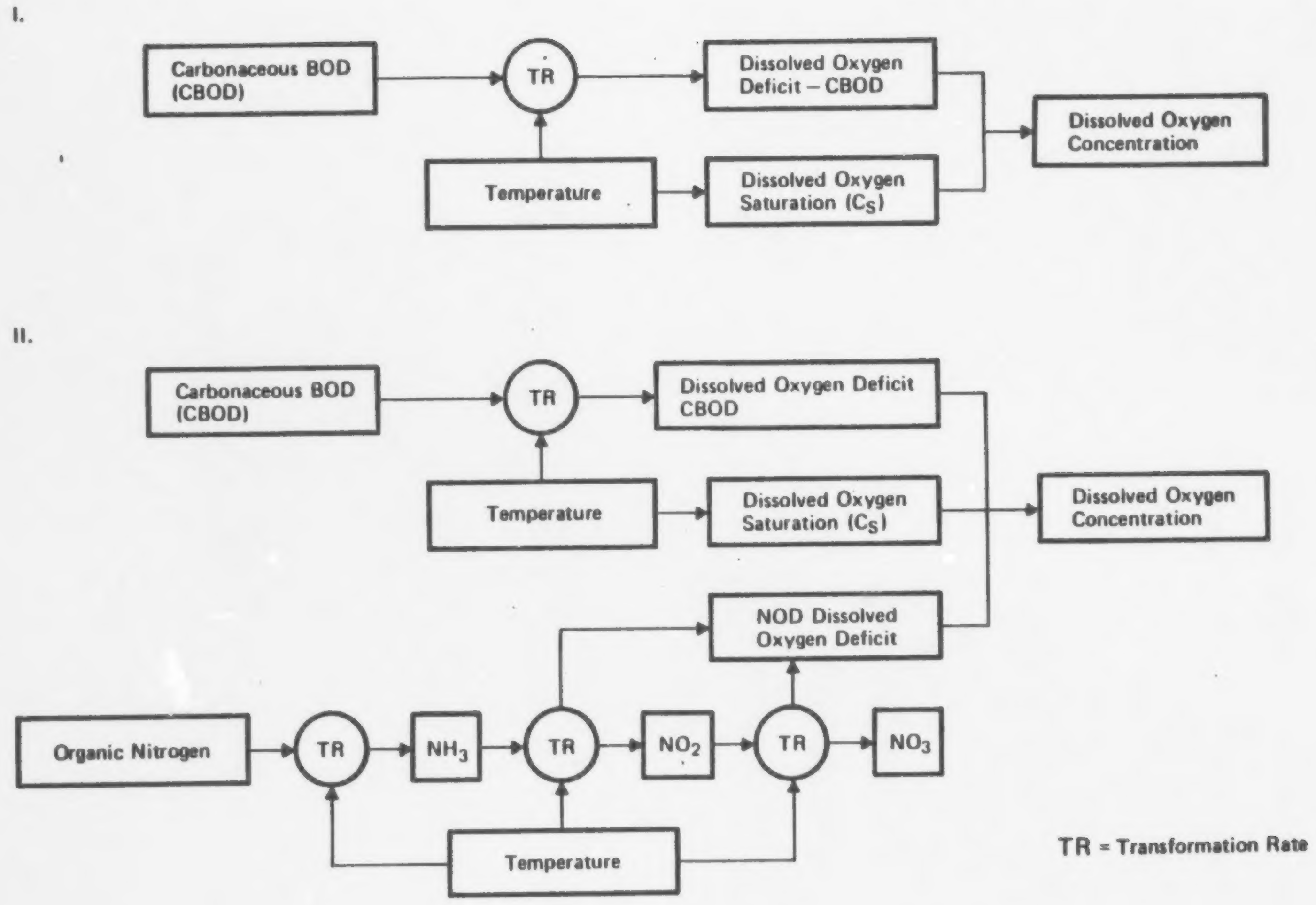

I.
Carbonaceous BOD (CBOD)
TR
Temperature
Dissolved Oxygen Deficit – CBOD
Dissolved Oxygen Saturation (C_S)
Dissolved Oxygen Concentration
II.
Carbonaceous BOD (CBOD)
TR
Temperature
Dissolved Oxygen Deficit CBOD
Dissolved Oxygen Saturation (C_S)
NOD Dissolved Oxygen Deficit
Dissolved Oxygen Concentration
Organic Nitrogen
TR
NH_3
TR
NO_2
TR
NO_3
Temperature
TR = Transformation Rate

Figure 3-1. Combinations of variables which can be considered for DO analysis in streams and rivers (continued).

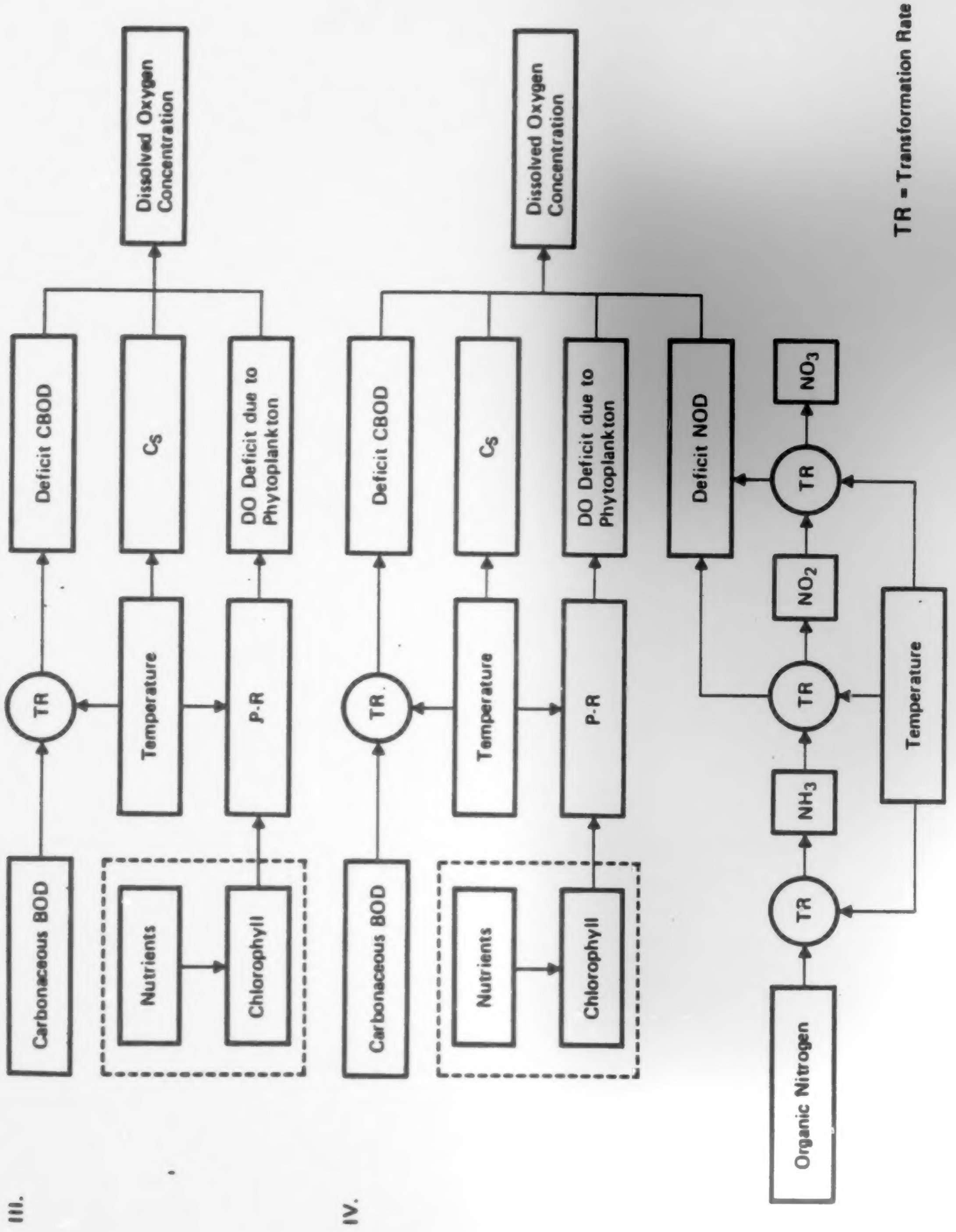

III.
Carbonaceous BOD
TR
Deficit CBOD
Temperature
C_S
Nutrients
Chlorophyll
P-R
DO Deficit due to Phytoplankton
Dissolved Oxygen Concentration
IV.
Carbonaceous BOD
TR
Deficit CBOD
Temperature
C_S
Nutrients
Chlorophyll
P-R
DO Deficit due to Phytoplankton
Deficit NOD
Organic Nitrogen
TR
NH_3
TR
NO_2
TR
NO_3
Temperature
Dissolved Oxygen Concentration
TR = Transformation Rate

Figure 3-1. (concluded).

calculations and relating the selected analysis framework to the decisions to be made at the conclusion of the project.

It should be pointed out that the illustrated reaction sequences in combinations II and IV show sequential oxidation of NH_3 to NO_2, which oxidizes to NO_3. In many situations the NO_2 concentration level observed and calculated is very low or tends to be uniform. Under these circumstances, the analysis could consider a single reaction sequence for ammonia oxidation, converting NH_3 directly to NO_3. It should also be noted that where algal problems are severe, NH_3 may be taken up directly by algae.

It will be important in many situations to distinguish between organic nitrogen and ammonia concentrations, rather than define the nitrogenous oxygen demand (NOD or NBOD) load on the basis of TKN concentrations, which are composed of both these forms. Organic nitrogen must first hydrolyze to ammonia before its oxygen demand will be exerted. Time and space lags in the resultant dissolved oxygen profile, due to this sequential reaction, may be significant. If the two species of nitrogen are combined in the calibration and verification effort, the apparent oxidation rate (K_n) will be lower than the actual first-order oxidation rate of ammonia. The ratio of TKN to NH_3-N affects the value of the overall oxidation rate. Where this ratio changes after treatment, the modeler is faced with additional uncertainty.

Several levels of analysis can be used for considering the influence of phytoplankton. These are summarized in Table 3-2. Level A, which uses measured values of P-R and diurnal dissolved oxygen data, may be satisfactory in many cases. When significant changes in nutrients or light extinction coefficient are anticipated, the level B analysis should be considered. Level C analysis

TABLE 3-2. METHODS OF ANALYSIS FOR PHYTOPLANKTON AND AQUATIC WEEDS

A. Measure P-R and/or Diurnal Swings in DO: Employ measured value in steady-state or quasi steady-state models

B. Measure Chlorophyll a, Light, Light Extinction, Nutrients: Employ the results in steady-state or quasi steady-state models

Calculate P-R

Compare to P-R Data and Diurnal Swings

C. Model Chlorophyll a, Nutrients, Dissolved Oxygen etc., with Calibration and Verification: A time-variable, nonlinear modeling framework is required.

increases the program costs for data and modeling by several orders and should be used when the problem is dominated by phytoplankton oxygen production and utilization and environmental or control costs are significant.

When aquatic weeds are the cause of diurnal fluctuations, the only analytical framework available is level A in Table 3-2. Other quantitative aquatic weed analysis frameworks are essentially in the research stage and do not appear to be appropriate for use in a decision-oriented project.

First-order rates employing sequential reactions generally have proven adequate for waste load allocation analysis. Most of the available computerized solution techniques employ these formulations. Therefore, the four combinations of variables defined in Figure 3-2, which are most frequently employed in dissolved oxygen analysis, can be characterized by first-order transformation functions (TR1, TR2, TR3, TR4, etc.). Ranges of the specific first-order rates for the various reactions are discussed in Section 3.3 as are the procedures for defining site-specific reaction rates for various levels of treatment. There are circumstances, particularly in systems with low

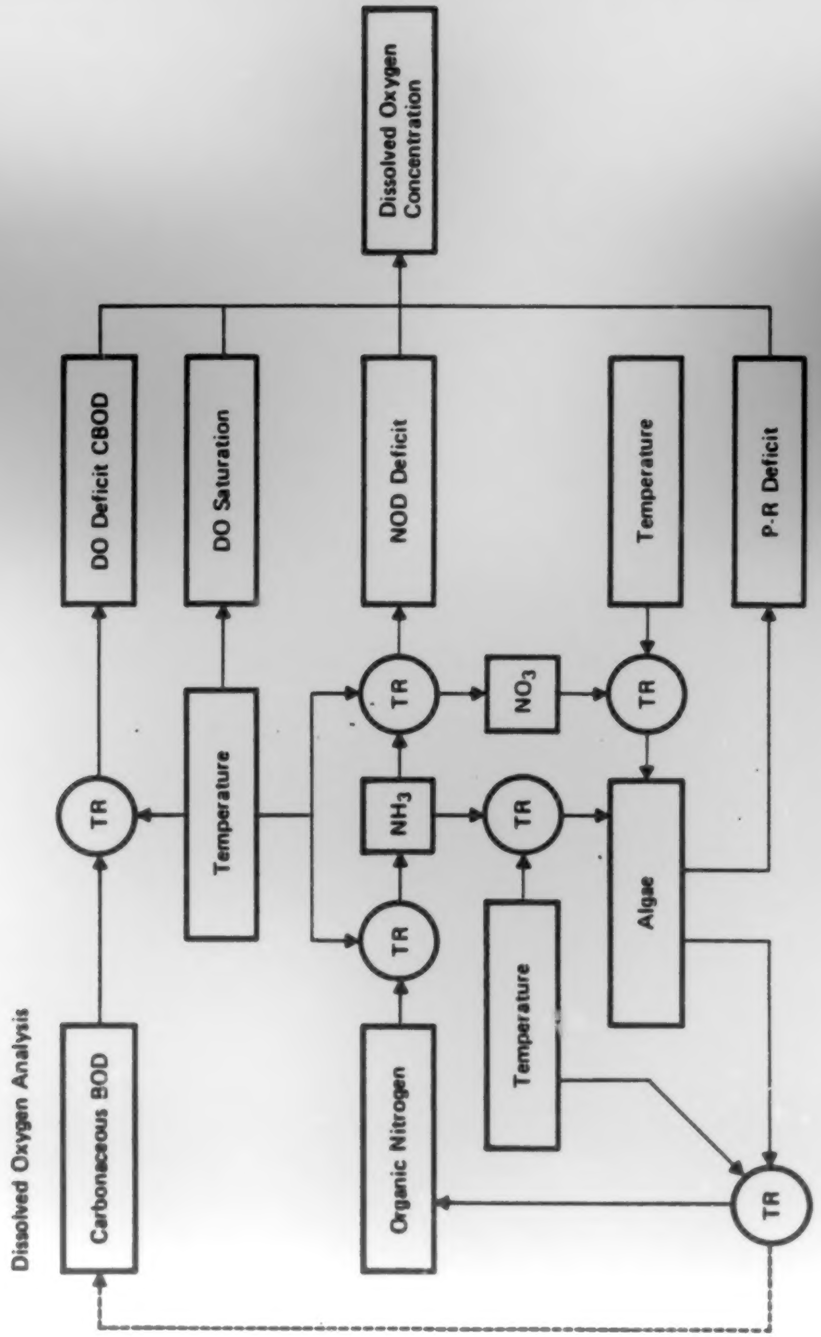

Dissolved Oxygen Analysis
Carbonaceous BOD
TR
Temperature
DO Deficit CBOD
DO Saturation
Organic Nitrogen
TR
NH_3
TR
NO_3
NOD Deficit
Temperature
TR
Algae
Temperature
P-R Deficit
TR
Dissolved Oxygen Concentration

Figure 3-2. Feedback reaction sequence.

dissolved oxygen, where nonlinear kinetic formulations can be considered. The nonlinear formulation usually employed is Michaelis-type kinetics in which the overall rate of reaction reduces as a chemical species is depleted. Dissolved oxygen is one of the chemical species that controls these types of kinetic formulations. In particular, the rate of nitrification has been shown (8, 9) to be controlled by dissolved oxygen levels at and below 2 mg/l. One method of approximating the nonlinear nitrification reactions has been to use lower values for first-order reaction rates in areas of low dissolved oxygen concentrations. Low dissolved oxygen concentrations can also reduce the rate of BOD oxidation, sediment oxygen utilization, and increase the release of contaminants from the benthos. These latter reactions are influenced only at very low dissolved oxygen levels such as 0.1 mg/l or lower. In bodies of water with large detention times, feedback reaction sequences have occasionally been employed (10). For example, the death and decomposition of algal cells returns organic nitrogen to the system. Feedback reactions can utilize first-order kinetics in dissolved oxygen analysis and have been used to model larger estuaries. The usual reaction sequence employed for dissolved oxygen investigation is shown in Figure 3-2. This feedback reaction sequence may be appropriate for larger river systems. In systems where dissolved oxygen levels are controlled by phytoplankton populations that are internally controlled by nutrients and available light, nonlinear phytoplankton kinetic models may be appropriate. In past applications these models usually have not been supported by the data base needed for calibration and verification and have employed literature values rather than site-specific reaction rate coefficients. While this type of kinetic structure can be found in available computational software, its use should be limited to dissolved oxygen problems that are controlled by phytoplankton levels and to nutrient removal decisions involving large costs. The data base required to make the analysis meaningful in decision making is very extensive and needs to be collected in support of studies using this modeling framework.

For dissolved oxygen analysis in streams and rivers, the basic modeling framework should consider sequential reactions with first-order kinetics, as illustrated in Figure 3-1. The necessity for departures from this norm should be documented at an early point in the project and should particularly address the additional information required in the decision-making process.

Guidelines for Selecting a Model. Selecting a model involves so many site-specific considerations that a detailed selection procedure that quantitatively ranks the acceptability of the model against selected criteria may be appropriate. An example of such a procedure is given in Reference 11. Herein, we have chosen to provide a series of practical guidelines that may assist in selection.

The guidelines fall under two categories: technical and operational. The technical guidelines previously discussed in this section ultimately are concerned with matching the model capabilities to the important physical and biochemical processes of the prototypical system. The operational guidelines are concerned with the ease and cost associated with model operation.

The following is the sequence of guidelines with a brief discussion of the considerations involved.

Technical Guideline #1: Determine Important Features of Prototypical System that are Required in the Analysis. The important mechanisms that generally govern the DO distribution in streams have been discussed in Section

2.2 and the means of identifying the site-specific analysis required are outlined in this section and in other publications (e.g., 11). With this background, site-specific data should be collected and reviewed to undefstand the system and establish the important factors associated with the identified problem (problem identification is discussed in Section 4.1). Valuable information can also be obtained from other experienced professionals, especially those who have modeling experience or site-specific field experience, and from personal site visits.

Technical Guideline #2: Review Available Models and Model Capabilities. Section 3.2 outlines the capabilities for several models selected primarily on the basis of professional acceptance, availability, and support. Many other models are available that contain the same capabilities as those presented here and should not be excluded from consideration.

It is important to be aware of those capabilities that involve a substantial increase in complexity. In simulating physical processes, a significant increase in complexity is associated with simulating time-variable hydraulics. This is because the model must solve the more complex equation of motion rather than, in the case of steady flow, the continuity equation. In simulating the biochemical processes, a significant increase in complexity is required to simulate (as opposed to prescribe) the photosynthetic process, water temperature, and nitrification.

Technical Guideline #3: Match Important Features of Prototypical System with Model Capabilities. An important step in model selection is comparing the important features of the prototypical system with the model capabilities and selecting, as techncially acceptable, those models whose capabilities match the features of the system. A rule of thumb is to select the simplest model(s) that retains all important features in the prototypical system. Choosing a more complex model is not cost effective since data requirements and computer cost tend to increase rapidly. An overly complex program will not usually result in an improved simulation and may increase uncertainty in the analysis.

Technical Guideline #4: Confirm Selection of Technically Acceptable Models. The information given in this report only permits a preliminary selection of a technically acceptable model(s). To confirm that the models are indeed technically appropriate, the potential user should consult the user's manual and other support documents, contact and discuss the potential application with members of the support agency, and consult with other experienced professionals.

Operational Guideline #1: Selection of Candidate Models Based on Ease of Application. Once a technically acceptable model has been selected, it is necessary to estimate the ease of applying it. However, it is very difficult to evaluate the adequacy of documentation and support and realistically estimate costs without prior experience with the model. Therefore, it is recommended that the support agency be consulted. It may be possible that special support arrangements (including short courses or informational or personnel exchanges) are available under existing intra- or interagency agreements or otherwise could be made available to the potential user. The support agency may also be able to provide the potential user with a list of local users who could be contacted for information regarding their past or current experience with the computer program associated with the model.

Operational Guideline #2: Selection of Candidate Models Based on Cost of Application and Problem Significance. It is difficult to estimate overall costs involved in a model application because each application differs in scope and complexity, and the ability to solve or avoid certain problems is very dependent on the experience and technical background of the analysts involved. However, machine requirements and costs associated with typical runs are usually estimated in the program documentation. As a rule, the simpler the model, the less expensive it is to apply. Again, it is essential that the support agency and other experienced professionals be contacted for information or assistance.

Once an estimate of the costs of application has been made, it should be compared with the benefits of using the program as part of the water quality modeling effort and the overall importance of the problem. In other words, the WLA study costs should be consistent with the economic, social, or environmental values associated with the problem and its solution.

Operational Guideline #3: Selection of Candidate Models Based on Data Availability and Data Acquisition Costs. All models require data for input, calibration, and verification. It is best if model selection is not restricted by availability of data and the decision is made to acquire the specific type of data required for the model. On the other hand, if data availability is a constraint, selection of a less sophisticated model than would be warranted on technical grounds may be appropriate.

Summary. The first step in model selection is to determine which programs are technically acceptable, based on an understanding of the important physical and biochemical processes in the prototypical system. The second step is to determine the ease and costs of application of those which are technically acceptable. The result of the second step is a list of candidate models which may or may not be ranked according to convenience and cost. The final selection of the preferred model from the list of candidates is based on the overall judgment of the potential user taking into account all of the factors discussed.

3.2 AVAILABLE MODELS AND MODEL FEATURES

In this section the term model, following commonly used terminology, is used to describe computer programs. However, strictly speaking, programs are not models until the user structures them with the geometry, hydrology, loading and reaction rate factors that are representative of the particular receiving water system being analyzed. It is only when this is done that the computer programs described can be considered to be mathematical models of the user's system.

Numerous models are available to analyze DO variations in a stream for a waste load allocation study. The models described in this section have been selected for discussion because:

- They are in the public domain.
- They are available at a minimal cost from various public agencies.
- They are supported to various extents by federal and/or state agencies. The form of support is generally telephone contact to a staff of engineers and programmers who have experience with the model and provide guidance usually free of charge.
- They have been used extensively for various purposes, particularly waste load allocation studies, and are generally accepted by the profession.

- They form a sequence of more technically complex models, i.e., each model takes into account additional phenomena and/or simulates a given phenomenon in a more detailed manner.

The selection procedure should not be limited to those models discussed in this document. Other computer programs (models) that are available to a project or organization should be given consideration. The discussions and criteria presented in this document can be employed as major elements in the selection process. One additional consideration in this process can be the experience and familiarity of the technical staff with a particular computer program.

It is suggested, however, that where project staffs do not have access to or familiarity with other computer programs, effort would be most effectively focused on the computer programs selected for discussion in this document. A brief description of the selected computer programs follows.

DOSAG-I is a program that solves the steady-state, one-dimensional equations that simulate the dissolved oxygen response in a stream network. The original model was developed by the Federal Water Pollution Control Administration (subsequently the US EPA Water Quality Office) and later modified by the Texas Water Development Board (12). The model solves the Streeter-Phelps equation, modified to include both carbonaceous and nitrogenous oxygen demands, for a series of uniform reaches that are assembled to simulate the stream network. Waste loads enter at the upstream ends of each reach. The model also can simulate the effects of two DO control options. An analyst can specify one of five treatment levels governing the waste load BOD and the model will estimate required flow augmentation to meet a specified DO standard.

SNSIM is a computer program that solves the steady-state, one-dimensional form of the stream equation. The code was developed by Robert E. Braster, Steven C. Chapra, and George A. Nossa of the US EPA Data Systems Branch in New York (13). The model solves the Streeter-Phelps equation, modified for NBOD and CBOD loads, for a series of uniform reaches that represent the stream network. Waste loads enter as point sources at the upstream end of the reach or as uniformly distributed nonpoint sources along the stream reach. The user may specify values for benthic oxygen demand and for net P-R from photosynthesis.

QUAL-II solves either the steady-state or time-variable stream equations. The program simulates CBOD, DO, temperature, chlorophyll a, phosphorus, NH_3, nitrate, nitrite, coliform bacteria, radioactive material, and any three conservative constituents in a stream network. It can be run in two modes: completely steady state, or dynamic in terms of water quality with steady-state hydraulics. The program takes into account longitudinal dispersion, sediment oxygen demand, CBOD settling, and nitrification. A version that simulates denitrification has been developed for EPA Region IV. Photosynthesis is based on a simplified nutrient-light-algal cycle. Steady-state or dynamic stream temperatures can also be simulated. QUAL-II was developed by Water Resource Engineers for the EPA (14) and further improved for the Southeast Michigan COG in a 208 Study (15). It is an adaption of QUAL-I developed by the Texas Water Development Board (16, 17).

RECEIV II solves the time-variable one or quasi two-dimensional (vertically well-mixed) stream or estuary equations. The program contains separate hydraulic and water quality subroutines. The hydraulic program solves the continuity equation and the one-dimensional equation of motion for a series of volumetric elements connected by hypothetical links that serve as flow paths. The following eleven constituents are considered: phosphorus, coliform bacteria, ammonia nitrogen, nitrite nitrogen, nitrate nitrogen, total nitrogen (assumed conservative), CBOD, chlorophyll a, DO, salinity (assumed conservative) and one metal ion. The program considers the effects of photosynthesis, nitrification, settling, and sediment demand on DO and is, therefore, similar in water quality simulation capability to QUAL-II. However, unlike QUAL-II, water temperature cannot be simulated. The code can also accommodate dynamic waste loads. Unlike the other three models previously discussed, RECEIV-II uses metric units. RECEIV-II, developed by Raytheon Company (9,18) under contract to the Water Planning Division, US EPA, is an adaption of the RECEIV module contained in the EPA Storm Water Management Model (SWMM) (19).

The models selected here represent the typical range available. However, on technical grounds, many other models are comparable indeed, some are related to those described. Other available computer programs can be generally grouped into one of the following categories:

- variants of the models discussed here
- proprietary models held by consulting firms
- models developed for research purposes

Most of the models selected are one in a sequence or family of versions originating from some source. Any member of this family tree may be a logical candidate for program selection. For example, a metric version of DOSAG-I, called DOSAGM, was developed for Sao Paulo, Brazil (20); and an updated version of DOSAG-I, referred to as DOSAG-III, was developed by Duke and Masch of Water Resource Engineers (21) to include many of the water quality features of QUAL-II.

Many proprietary models are held by consulting firms and some individual consultants. One of the better known models in this category is the

Hydrocomp Simulation Model HPSF developed by Hydrocomp, International, Palo Alto, California (22). The quality portion of the model simulates temperature, BOD, coliform bacteria, zooplankton, sediment oxygen demand, organic nitrogen, DO, TDS, nutrients, and conservative constituents. The model also can be run continuously and the output can be frequency analyzed to determine, for example, the probability distribution of dissolved oxygen in the stream. A nonproprietary version of this program, the Hydrologic Simulation Model (FORTRAN [HSPF]) is now available from EPA Athens Environmental Research Laboratory (Ambrose, personal communication 1980). These programs are quite complex and require well-trained analysts. Another proprietary model is Raytheon's RECEIV III, an updated version of RECEIV-II, which, unlike its predecessors, includes longitudinal dispersion as well as other effects.

Numerous analytical frameworks cited in the technical literature have been developed for research purposes to stimulate DO. These models either have been developed from scratch or are modified versions of existing models. For example, Wu and Ahlert (23) discussed a steady-state BOD, NH_3-N, DO model that takes into account photosynthesis, sedimentation, and nonpoint source distributed loads; Lin, Fan & Erickson (24) included the effects of transient instream temperature in a program that solves the DO and BOD equations; Novotny and Krenkel (25) modified DOSAG-I to include settling and benthic demand; and Sparr (21) applied QUAL-I to the lower Mississippi to investigate the effect of longitudinal dispersion.

Salient features of the models selected for discussion in this manual are summarized in Tables 3-3 through 3-15. References 27 and 28 describe many other available water quality models. The categories and subject of tables, modified from Evaluation of Water Quality Models: A Management Guide for Planners (11), are as follows:

TABLE 3-3. CAPABILITIES: TEMPORAL AND SPATIAL FEATURES

Model	Time Variability	Spatial Dimensions	Event or Continuous
DOSAG-I	Steady State[1]	One Dimensional	Not Applicable[2]
SNSIM	Steady State	One Dimensional	Not Applicable[2]
QUAL-II	Steady State or Dynamic Water Quality	One Dimensional	Not Applicable[2]
RECEIV-II	Dynamic	One or Quasi-Two Dimensional	Event or Continuous

[1] Set up but not restricted to simulate mean monthly conditions.
[2] Any steady state (usually low flow) condition.

TABLE 3-4. CAPABILITIES: HYDRAULIC FEATURES

Model	Receiving Water Type	Single Reach or Network	Other Features
DOSAG-I	Stream	Both	
SNSIM	Stream	Both	
QUAL-II	Stream or Completely Mixed Reservoir	Both	
RECEIV-II	Stream or Completely Mixed Estuary	Both	Considers wind-stress, control structures (dam), tidal boundary condition

TABLE 3-5. CAPABILITIES: WASTE LOADS[1], SINKS, AND SOURCES OF DO

Model	Waste Load Characteristics			Dissolved Oxygen		Special Features and/or Limitations
	Types	Single or Multiple	Loading Rate	Sinks	Sources	
DOSAG-I	Point	Multiple	Constant	CBOD NBOD	Reaeration	Can specify treatment levels; model will estimate required flow augmentation to meet specified DO standard.
SNSIM	Point & Nonpoint (distributed)	Multiple	Constant	CBOD NBOD benthal demand, Algal respiration	Reaeration, photosynthesis (specify P-R)	
QUAL-II	Point & Nonpoint	Multiple	Constant	CBOD NBOD benthal demand, Algal respiration	Reaeration, photosynthesis (nutrient-algal)	Allows for settling of BOD material, will estimate required flow augmentation to meet specified DO standard.
RECEIV-II	Point	Multiple	Constant or Variable	CBOD NBOD benthal demand, Algal respiration	Reaeration photosynthesis (nutrient-algal)	Allows for settling of nitrate

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[1] Commonly refer to municipal and industrial waste discharges but may also apply to tributary or upstream inflows and agricultural or urban runoff.

TABLE 3-6. CAPABILITIES: CONSTITUENTS

Model	Modeled	Specified
DOSAG-I	DO, NBOD, CBOD (conservative constituents via zero reaction rate)	Temperature
SNSIM	DO, NBOD, CBOD (conservative constituents via zero reaction rate)	Temperature
QUAL-II	DO, CBOD, temperature, chlorophyll a, phosphorus, NH_3, nitrate, nitrite, coliform bacteria, radioactive material, and any three conservative constituents	
RECEIV-II	DO, CBOD, chlorophyll a, phosphorus, NH_3, nitrate, nitrite, total nitrogen, coliform bacteria, salinity & one metal ion.	Temperature

TABLE 3-7. CAPABILITIES: PHYSICAL AND BIOCHEMICAL PROCESSES SIMULATED

Model	Physical Processes	Biochemical Processes
DOSAG-I	Advection, dilution, reaeration	1st-order decay of CBOD, NBOD; coupled DO with CBOD and NBOD
SNSIM	Advection, dilution, reaeration	1st-order decay of CBOD, NBOD; coupled DO with CBOD and NBOD, benthic demand, net photosynthesis (specified)
QUAL-II	Advection, dilution, reaeration, longitudinal dispersion, stream heat balance	1st-order decay of CBOD, CBOD removal by deposition; coupled DO with CBOD, benthal demand, photosynthesis and nitrification (based on nutrient-algal model)
RECEIV-II	Advection, dilution, reaeration	1st-order decay of CBOD; coupled DO with CBOD, benthal demand, photosynthesis & nitrification (based on nutrient algal model)

TABLE 3-8. MODEL CAPABILITIES: REAERATION FORMULATIONS

Model	Options
DOSAG-I	Four options: input directly; calculated as a function of velocity and depth following Churchill et al., Langbein & Durum, O'Connor & Dobbins, or Owens & Gibbs; calculated as a function of flow; method of Thackston & Krenkel
SNSIM	Three options: input directly; calculated as a function of velocity and depth as above; method of Tsivoglou et al.
QUAL-II	Multiple options similar to DOSAG-I, including Tsivoglou's equation.
RECEIV-II	Churchill et al.

TABLE 3-9. ACCURACY: PRINCIPAL ASSUMPTIONS

Model	Quantity	Quality
DOSAG-I	Assumes steady state; geometry and velocity are uniform throughout reach; no lateral or vertical variation in velocity; velocity and depth can be expressed as power functions of flow, completely mixed.	Assumes 1st-order decay of NBOD and CBOD (both specified), constant waste loads, neglects benthic demand and photosynthesis; reaction rates constant in a reach; well mixed laterally and vertically, no longitudinal dispersion.
SNSIM	Assumes steady state; geometry and velocity are uniform throughout reach; no lateral or vertical variation in velocity; velocity and depth can be expressed as power functions of flow, completely mixed.	Assumes 1st-order decay of NBOD and CBOD (both specified); constant waste loads; constant benthal and photosynthetic demand; reaction rates constant in a reach; well mixed laterally and vertically; no longitudinal dispersion.
QUAL-II	Assumes steady state; geometry and velocity are uniform in a reach; no lateral or vertical variation in velocity; velocity and depth can be expressed as power functions of flow, completely mixed.	Assumes CBOD 1st-order decay; well mixed laterally and vertically; includes effects of benthic demand (specified), algal production (modeled), and nitrification (modeled) on oxygen; allows for CBOD settling (specified); includes longitudinal dispersion.
RECEIV-II	Assumes receiving water can be broken down into a system of completely mixed volumetric units called nodes connected by a series of links along which flows occur; no lateral variation in velocity.	Assumes CBOD 1st-order decay; includes effects of benthic demands (specified), algal production (modeled), & nitrification (modeled) on oxygen; model volume is well mixed; no longitudinal dispersion.

TABLE 3-10. DATA REQUIREMENTS: INPUT

Model	Geometric	Meteorologic	Hydrologic	Hydraulic	Water Quality	Effluent	Decay Rates	Other
DOSAG-I	Stream lengths, uniform reaches, connection scheme, coefficients for depth-flow regression	-	Headwater and tributary inflows, withdrawals	Coefficients for velocity-flow regression	Inflow concentration, and stream temperature	Flow rates and concentration	Reaeration and reaction rate coefficients, and temperature correction factors	Treatment factors, target minimum DO concentration
SHSIM	Stream lengths, uniform reaches, connection scheme, coefficients for depth-flow regression	-	Headwater and tributary inflows, withdrawals	Coefficient for velocity-flow regression	Inflow concentration, stream temperature, benthic demand, and photosynthetic demand	Flow rates and concentration	Reaeration and reaction rate coefficients, and temperature correction factors	-
QUAL-II	Stream lengths, connection scheme, length of computational element, coefficients for depth-flow regression	Cloud cover, atmospheric pressure, dry and wet bulb temperature, wind, evaporation coefficients	Headwater and tributary inflows, withdrawals	Coefficients for velocity-flow regression, bottom roughness (Manning's n)	Inflow concentration	Flow rates and concentration	Reaeration, reaction and settling rate coefficients, and temperature correction factors	Latitude, longitude and elevation of basin, day of year, computational time-step, target minimum DO concentration and sources of flow augmentation
RECEIV-II	Bathymetry to provide surface areas, and depths, channel lengths, widths, and depths	Rainfall intensity, evaporation rate, wind speed and direction	Headwater and tributary inflows	Roughness coefficient (Manning's n), initial velocities and depths, downstream control (weir)	Inflow concentration, water temperature, and initial conditions	Flow rates and concentration	Reaeration, reaction, and settling rate coefficients, and temperature correction factors	Hydraulic and quality timesteps

Note: Meteorological input is optional.

TABLE 3-11. DATA REQUIREMENTS: CALIBRATION AND VERIFICATION

Model	Hydrologic	Hydrodynamic	Water Quality
DOSAG-I	Stream flow	Stream velocity	Concentrations
SNSIM	Stream flow	Stream velocity	Concentrations
QUAL-II	Stream flow	Stream velocity	Concentrations and temperature
RECEIV-II	Stream flow	Stream velocity, depth	Concentrations

TABLE 3-12. EASE OF APPLICATION: OUTPUT FORM AND CONTENT

Model	Output Form	Output Content
DOSAG-I	Computer printout	a) Listing of input data b) DO, CBOD and NBOD concentrations at start and end of each reach, and magnitude and location of minimum DO concentration in each reach
SNSIM	Computer printout	a) Listing of input data b) Variation of DO, CBOD and NBOD concentrations and DO deficit and source of deficit along stream reach
QUAL-II	Computer printout	a) Listing of input data b) Concentrations and temperature by reach and computational elements at specified time steps c) Maximum, minimum and average concentration, temperature, flow, velocity and depth for each reach
RECEIV-II	Computer printout, velocities also written on scratch tape/disk for quality mode (optional tape/disk storage of results for plotting)	a) Input data listing b) Channel flows and velocities, junction depths and concentration of constituents at prescribed time intervals

TABLE 3-13. EASE OF APPLICATION: SOURCES, SUPPORT AND DOCUMENTATION

lel	Source(s) of Model	Source(s) of Support	Nature of Support	Documentation (Reference #)	Source(s) of Documentation	Adequacy of Documentation
AG-I	EPA Planning Assistance Branch, Wash. D.C.	-	-	12	NTIS PB 202 974	Adequate theory and user's manual, good input data format
	Data Systems Section Texas Dept. of Water Resources, P.O. Box 13087 Austin, Texas 78711	Same	Telephone contact			
IM	EPA Data Systems Branch, 26 Federal Plaza New York, N.Y. 10007	Same	Telephone contact	13	NTIS PB 241 923	Adequate theory and user's manual, well-organized input data format
L-II	Center for Water Quality Modeling U.S. Environmental Protection Agency Athens, GA 30613 (404) 546-3585	-	Software Maintenance, Workshop, Technical Assistance through Official EPA Channels	14		Good discussion of theory, and assumptions Adequate user's manual, includes good input data information
	Texas Dept. of Water Resources (see above)	Same	Telephone contact			
EIV-	EPA Planning Assistance Branch Wash. D.C. 20460	Same	Telephone contact	18	EPA Planning Assistance Branch Wash. D.C. 20460	Excellent description of input data needs; well-organized input format
	Raytheon Company P.O. B[illegible] 360 Ports[illegible]h, R.I. 02871	Same	Telephone contact			

TABLE 3-14. EASE OF APPLICATION: EQUIPMENT AND PROGRAMMING REQUIREMENTS

Model	Requirements
DOSAG-I	Requires any computer with about 27,000 word storage and a FORTRAN-IV compiler. No tapes or disks needed.
SNSIM	Written in FORTRAN IV, version SNSIM-1 compatible with IBM 370/155; SNSIM-2 compatible with IBM 1130. Storage requirements similar to DOSAG-I. No tapes or disks needed.
QUAL-II	Written in FORTRAN IV (level G) to be compatible on UNIVAC 1108, CDC 6400, and RCA Spectra 70/45 and therefore is almost machine independent; requires 45,000 word storage. No tapes or disks needed.
RECEIV-II	Written in FORTRAN IV (level G)[a] to be compatible on IBM 370/155 and Honeywell 6000/60; requires approximately 50,000 words of storage; tapes or disks may be necessary depending on application.

[a] Also compatible to ANSI Standard FORTRAN.

TABLE 3-15. OPERATING COSTS

Model	Acquisition	Machine Costs[a,b] Per Run (dollars)	Labor Costs (person weeks)		
			Set Up[c]	Running[a]	Analysis[a]
DOSAG-I	Nominal	1-5	2-6	Negligible	1
SNSIM	Nominal	1-5	2-6	Negligible	1
QUAL-II	Nominal	1-5 or less	2-10	Negligible	1[d]
RECEIV-II	Nominal	15-100 (quantity) 10-50 (quality)	5-20	Routine runs take minimal time	The complexity of model requires at leas several hours analysis for e run to be eval ated

[a] For each run, including those used for calibration, verification and projections.

[b] Approximate range for a single run on a typical application using a commercial IBM 370/55 during daytime hours. These costs are dependent on many factors and should be used in a relative sense.

[c] Set-up time for each model depends on the complexity of the application, the form of the available data, and staff capabilities.

[d] Runs including nutrient-algal simulation initially require at least several hours.

Partial Source: Reference (11).

	Table
Capabilities:	
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Information presented under the first three table subjects (Capabilities, Assumptions, and Data Requirements) is primarily technical and required to evaluate whether the model simulates the important physical and biochemical features of the problem. Information presented under the table subjects, Ease of Application and Operating Costs, is primarily nontechnical or related to operational features of the models. This information is needed to evaluate the cost associated with and the ease of acquiring the model, getting the model running on your system, calibrating and verifying the model, and finally applying the model.

The information provided in these tables is primarily qualitative and sufficient to determine whether a model may be suitable for a particular

TABLE 3-16. SUMMARY OF FEATURES

	Time Scale	Space Scale	Event or Continuous	Water Body	Types of Loads Accepted	Loading Rate Accepted	Water Quality Parameter Modeled	Processes Simulated		Calibration/ Verification Parameters
								Chemical/Biolgoical	Physical	
DOSAG-I	steady state	1	-	stream network	multiple point source	constant	DO, CBOD, NBOD conservative[a]	1st-order decay of NBOD, CBOD coupled DO	dilution, advection, reaeration	Q, V, C
SNSIM	steady state	1	-	stream network	multiple point source & distributed source	constant	DO, CBOD, NBOD conservative[a]	1st-order decay of NBOC, CBOD, coupled DO, benthic demand (s), photosynthesis (s)	dilution, advection, reaeration	Q, V, C
QUAL-II	steady state hydraulic, steady & dynamic quality	1	-	stream network	multiple point source & nonpoint source	constant or time variable	DO, CBOD, temperature, ammonia, nitrate, nitrite, algae, phosphorus coliform, radioactive substances, three conservative substances	1st-order decay of NBOD, CBOD, coupled DO, benthic demand (s), CBOD settling (s), nutrient-algal cycle	dilution, advection, reaeration, heat balance	Q, V, C
RECEIV-II	dynamic	1,2	event or continuous	stream network or well-mixed estuary	multiple point source	constant or time variable	DO, CBOD, ammonia, nitrate, nitrite, total nitrogen, phosphorous coliform, algae, salinity, one metal ion	1st-order decay of CBOB, coupled DO benthic demand (s), CBOD settling (s), nutrient-algal cycle	dilution, advection, reaeration	Q, V, C

[a] Use CBOD or NBOD with zero or low decay rates

(s) = specified

TABLE 3-17. HEIRARCHY OF MODELS BASED ON SELECTED FEATURES

Model	Multiple Constant Point Sources of		Distributed Sources of BOD	Benthal Demand	Algal Production	Longitudinal Dispersion	BOD Settling	Time-variable Waste Loads (and quality)	Time-variable Flow
	NBOD	CBOD							
DOSAG-I	X	X	X^c				X^d		
SNSIM	X	X	X	X^a	X^a		X^d		
QUAL-II	X	X	X	X^a	X^b	X	X^a	X^e	
RECEIV-II	X	X	X^c	X^a	X^b		X^d	X	X

[a] Specified (i.e., input to the model)
[b] Simulated in a nutrient-algal cycle
[c] Can be simulated approximately by input of load at beginning of each multiple segment.
[d] Can be simulated by making $K_r > K_a$
[e] Meteorology only

application. For some of the models, more quantitative information is given in Reference (11). For complete information the potential user must consult the appropriate user's manuals and other supporting documentation. The Center for Water Quality Modeling, EPA Environmental Research Laboratory, Athens, Georgia (Mr. Thomas O. Barnwell) is a source of information and technical support. A brief description of the contents of each of these tables follows.

Table 3-3 Model Capabilities: Temporal and Spatial Features. The difference between static and dynamic models, single versus multidimensional models and the significance of various time and space scales in the context of the water quality problem have been discussed in Section 3.1. The models considered in Table 3-3 are all deterministic, one or two dimensional, and can calculate steady-state or dynamic solutions. RECEIV-II may be run either in (1) an event mode (i.e., the simulation period is limited to the duration of a relatively short-lived event of interest, e.g., a storm, a diurnal or weekly pattern of waste load discharge, or a weekly period of changing stream flow) or in (2) a continuous mode in which numerous events over larger periods (e.g., monthly, seasonal, or annual) are simulated. The operational difference is associated with the time step used in the calculation and the associated requirements for data on flow, loads, etc. as a function of time. Event simulations require data at smaller time steps than continuous simulations.

Table 3-4 Model Capabilities: Hydraulic Features. All of the models selected are appropriate for a stream network, i.e., a system of individual stream segments connected together. Additional capability is found in the RECEIV-II model, which may be applied to well-mixed estuaries. The first

three models use inputted upstream, tributary and wasteload flows to compute downstream flows assuming steady state. RECEIV-II simulates time-variable hydrodynamics and, therefore, requires downstream depth-flow control. In RECEIV-II the control is a dam or a tidal boundary condition.

Table 3-5 Model Capabilities: Waste Loads, Sinks, and Sources of DO. The models vary in ability to simulate point versus nonpoint source waste loads, constant or time-variant loading (or discharge) rate and DO demands from benthal deposits, photosynthesis, and nitrification. QUAL-II also allows for BOD settling. In most cases if a model does not permit distributed loads, such loads can be approximated by a series of point sources.

Table 3-6 Model Capabilities: Constituents. The models vary significantly in terms of the number and type of constituents for which calculations are performed. The number of constituents analyzed usually reflects the number and complexity of biochemical processes simulated, and is shown in Table 3-7. In the more complex programs, i.e., QUAL-II and RECEIV-II, provision is made for selecting only those constituents (and therefore processes) of interest. Therefore, it is relatively easy to "zero out" constituents that are not applicable. Sometimes it is possible to change the number or type of constituents simulated by minor modifications in the code. For example, conservative constituents may be simulated by setting reaction rates to zero; or direct oxidation of NH_3 to NO_3 may be simulated by assigning a large value to the NO_2 to NO_3 oxidation rate coefficient.

Table 3-7 Model Capabilities: Physical and Biochemical Processes Simulated. The physical processes simulated are usually advection, dilution,

and reaeration since these are considered essential to describe most physically related DO variations. Settling may also be simulated directly, or by adjustment of rates such as K_r and K_d. The biochemical processes simulated are quite varied and can be classified in the following increasing order of complexity:

- BOD-DO coupling (usually first-order reactions)
- NBOD-CBOD-DO linkage (i.e., inclusion of nitrogenous biochemical oxygen demand in DO balance--usually first-ordered reactions)
- Prescribed sediment oxygen demand and/or photosynthetic related oxygen production and demand (usually first-order reactions)
- Simulated nutrient-algal-dissolved oxygen linkage (usually nonlinear reactions)

Sometimes it is possible to consider certain effects even though the effects are not explicitly contained in the program. For example, if the program makes a distinction between the BOD decay rate (K_r) and the oxidation rate associated with BOD decay (K_d) [see Section 2.3] then settling can be considered by making K_r greater than K_d. If the program does not make such a distinction, it may be relatively easy to modify the code accordingly.

Table 3-8 Capabilities: Reaeration Formulations. Most models permit direct input of the reaeration coefficient or selection from several commonly used correlations or methods.

Table 3-9 Accuracy: Principal Assumption. Each model makes a number of assumptions. The simpler computer program simulation techniques, by neglecting certain effects, assume that such effects are not important for the case being considered. The more complex models, although they may consider additional effects, also incorporate assumptions in mathematically representing the prototypical system.

In the context of water quality modeling, two components should be considered in judging the accuracy of computer programs. The first consideration deals with the accuracy of the numerical solution technique for the differential equations representing water quality employed by the program, as compared to analytical solutions of the same differential equations. The second consideration is concerned with the adequacy and accuracy of the formulations of rates and transport that are employed in the differential equation that is being used in the analysis. In most cases, the computer programs can be operated to provide adequate accuracy for decision making, considering both of the components indicated above.

Table 3-10 Model Data Requirements: Input. The input data requirements increase with the complexity of the flow and quality mathematical formulations. The first three models assume steady-state hydraulics formulae, which then require specification of regression coefficients (see equations 2-3 through 2-5) to estimate velocity and depth required in the reaeration formulae. The more complex models such as RECEIV-II solve a form of the momentum equation, which requires more detailed characterization of the stream geometry and roughness. Similarly, the data required to simulate the nonlinear nutrient-algal-DO linkage is extensive.

Table 3-11 Data Requirements: Calibration and Verification. The type and amount of data required for calibration and verification increases as the complexity of the computer program becomes greater. Data requirements tend to increase rapidly with increasing complexity of analysis. Dynamic models require that data be obtained synoptically at a number of stations and over relatively short time intervals (e.g., minutes to hours).

Table 3-12 Ease of Application: Output Form and Content. All of the computer programs print results of the simulation and the input data. The more complex ones require scratch disks or tapes for storing intermediate results to be read subsequently in submodels or for storing information to be plotted.

Table 3-13 Ease of Application: Sources, Support, and Documentation. Two of the most important factors in facilitating the use of a new model are the adequacy of the documentation and the adequacy of the support available. The documentation should state the theory and assumptions in adequate detail, describe the program organization, and clearly present the input data requirements and format. A well-organized input data scheme is essential. The support provided by the support agency should include user access via telephone to programmers and engineers familiar with the model.

Table 3-14 Ease of Application: Equipment and Programming Requirements. All models are written in FORTRAN IV and most are machine independent. Storage requirements increase with program complexity.

Table 3-15 Operating Costs. Computer costs and labor requirements vary significantly depending on numerous factors as discussed in the notes at the bottom of Table 3-15. Therefore, these results are only to be used for comparisons between models.

Table 3-16 Summary of Features. For initial screening purposes, a summary of model capabilities is given in Table 3-16. For more detail the reader may then refer to Tables 3-3 through 3-8.

Table 3-17 Hierarchy of Models Based on Selected Features. To assist in initial model selection, Table 3-17 shows a hierarchy of models based on

important distinguishing features. As shown in this table, the programs increase in complexity. One of these programs should be adequate for most waste load allocation studies and, in general, the simpler program should be chosen if it contains all the features needed to simulate the important processes in the prototype. On the other hand use of a more complex model may be justified. Often, a complex model can be used with no more additional effort than a simple model by "turning off" processes. This procedure allows easy upgrading of the model as more information becomes available. QUAL-II, for example, can be used at the same analysis level as DOSAG-I and SNSIM, and requires no additional information.

3.3 MODELING PROCEDURES

Objectives of Site-specific Modeling

Assume that the water quality problem has been identified, the components of a water quality model have been defined, and a computational framework, such as a computer program, selected. The next task is to develop a site-specific water quality model that is directed towards each of the following objectives in the context of the water quality problem:

- Confirm that water quality problems do or will exist.
- Develop a quantitative understanding of system response to each significant load, source, and sink.
- Develop a quantitative understanding of system response to important constituents in the loads.
- Define the level of uncertainty in the quantitative understanding of system response.

The modeling work is directed towards defining the quantitative effects of the components that contribute to water quality problems. Critical examination of the residual uncertainties in the water quality analysis is usually of prime importance in the engineering and decision-making process. Therefore, it is necessary to include in the calibration and verification program a series of calculations and data sets that permits the analyst to check and cross-check the water quality effects ascribed to individual loads and constituents. In this context, the following questions are typical of those that will be helpful in setting up and evaluating the study program:

- How can the effect of two sources be differentiated using data and calculations?
- How can the effect of sediment oxygen demand, nonpoint sources, and point sources be differentiated?
- What data and calculations can be developed that will provide an assessment of the differences in reaction rates (such as BOD oxidation rates) between various sources?
- Is nitrification occurring in the system, and should it be anticipated under future conditions?
- What is the magnitude of dissolved oxygen fluctuations that cannot be accounted for by the present analysis?
- How do these fluctuations vary with time, seasonally and spatially?

In summary, the goal of the water quality analysis is to obtain an assessment of system behavior that will support decision making. Each study has unique requirements that may modify the modeling procedures suggested below; however, they are presented as a basis for meeting many project requirements and as a point of departure for development of individualized site-specific study programs.

Verify Model Calculations

The first step in any modeling analysis is to verify that the calculation technique to be used is functioning correctly. This can be accomplished by employing analytical solutions for specific simplified representations of the individual components to be included in the analysis effort. Table 3-18 is reproduced from the Areawide Assessment Procedures Manual (29) and contains analytical solutions to the stream dissolved oxygen equations. That manual also provides example calculations illustrating the use of these equations. Equation 3-1 can be employed to check the more complex computerized solutions to the dissolved oxygen equation in streams. Note that in equation 3-1, the convention for concentrations at x=0 (D_o, L_o, N_o) reflects concentrations after mixing with the waste load, in contrast to the convention employed in Table 3-18, in which the subscript "o" represents concentrations in the stream immediately upstream of the waste load.

$$D = + D_o e^{-j_a \phi(x)} \qquad \text{3-1(a)}$$

$$+ F_1 \left[e^{-j_r \phi(x)} - e^{-j_a \phi(x)} \right] \qquad \text{(b)}$$

$$+ F_n \left[e^{-j_n \phi(x)} - e^{-j_a \phi(x)} \right] \qquad \text{(c)}$$

$$+ \frac{S}{K_a H} \left[1 - e^{-j_a \phi(x)} \right] \qquad \text{(d)}$$

$$+ \frac{R - P_m/\pi}{K_a} \left[1 - e^{-j_a \phi(x)} \right] - P_m f(x,t) \qquad \text{(e)}$$

in which

$$F_1 = \frac{K_d L_o}{K_a - K_r} \qquad F_n = \frac{K_n N_o}{K_a - K_n} \qquad j_{r,n,a} = \frac{K_{r,n,a}}{U_o}$$

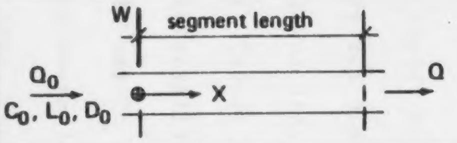
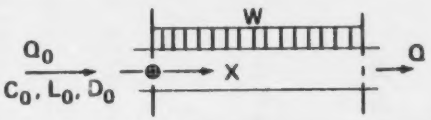
$$f(x,t) = \frac{1}{2(K_a{}^2 + \pi^2/p^2)}$$

$$\left\{ \left[K_a \sin \pi \frac{t}{p} - \frac{\pi}{p} \cos \pi \frac{t}{p} \right] - \left[K_a \sin \frac{\pi}{p}(t - \frac{x}{U}) - \frac{\pi}{p} \cos \frac{\pi}{p}(t - \frac{x}{U}) \right] e^{-j_a \phi(x)} \right\}$$

$$- \frac{2}{3\pi(K_a{}^2 + 4\pi^2/p^2)}$$

$$\left\{ \left[K_a \cos 2\pi \frac{t}{p} + \frac{2\pi}{p} \sin 2\pi \frac{t}{p} \right] - \left[K_a \cos 2\pi \frac{t}{p} + \frac{2\pi}{p} \sin \frac{2\pi}{p}(t - \frac{x}{U}) \right] e^{-j_a \phi(x)} \right\}$$

TABLE 3-18. SUMMARY OF SOLUTIONS FOR POLLUTANT CONCENTRATIONS IN THE RECEIVING WATERS

	Point Source	Distributed Source
		
Conservative C	$C = C_0 + W/Q$	$C = C_0 + wX/Q$
Reactive L	$L = L_0 e^{-K_r X/U} + (W/Q) e^{-K_r X/U}$	$L = L_0 e^{-K_r X/U} + \frac{w}{A \cdot K_r} (1 - e^{-K_r X/U})$
Coupled D	$D = D_0 e^{-K_a X/U}$ $+ L_0 \cdot \frac{K_d}{K_a - K_r} [e^{-K_r X/U} - e^{-K_a X/U}]$ $+ (W/Q) \cdot \frac{K_d}{K_a - K_r} [e^{-K_r X/U} - e^{-K_a X/U}]$	$D = D_0 e^{-K_a X/U}$ $+ L_0 \cdot \frac{K_d}{K_a - K_r} \cdot [e^{-K_r X/U} - e^{-K_a X/U}]$ $+ \frac{w}{AK_r} \cdot \frac{K_d}{K_a - K_r} [\frac{K_r}{K_a} e^{-K_a X/U} - e^{-K_r X/U} + \frac{K_a - K_r}{K_a}]$

NOTE:

Q = Flow
X = Distance
C = Conservative Substance Concentration
L = Reactive Substance Concentration (BOD)
D = Coupled Substance Concentration (DO Deficit)
U = Velocity
A = Cross-sectional Area

K_r = BOD Removal Coefficient
K_d = BOD Oxidation Coefficient
K_a = DO Reaeration Coefficient
C_0, L_0, D_0 = Concentration at X = 0
W = Point Surface Loading Rate
w = Non-point Source Loading Rate

This equation calculates the dissolved oxygen deficit distribution in space for:

- Initial deficit from upstream and/or waste flow - Equation 3-1a.
- BOD input from a waste load - Equation 3-1b.
- NH_3 oxidation from a waste load - Equation 3-1c.
- Sediment oxygen demand - Equation 3-1d.
- Phytoplankton oxygen production and respiration - Equation 3-1e.

The recommended approach is to use constant geometry and temperature (other than 20°C) in a simplified representation of a site-specific system and then to compare the dissolved oxygen deficit as calculated by the computer program and by the equations. Do this either for all loads, or preferably individually for each type of load, source, and sink. All computer codes for stream analysis can yield the dissolved oxygen deficit as output. This may require some programming effort, depending on the program version. The effort will prove a worthwhile investment in time both for this activity and for subsequent requirements to examine unit responses. It should be noted that Equation 3-1 can be employed to calculate the time-variable (quasi-steady state) dissolved oxygen response due to phytoplankton oxygen production and utilization. Therefore, this equation can be employed as partial confirmation of the dissolved oxygen calculations in the more complex time-variable phytoplankton models for streams. The approach is to run the time-variable model to steady state and, using the documentation, calculate P_m and R, internally generated by the model. These values can then be substituted into Equation 3-1. The stream equations in Table 3-18 and Equation 3-1 can be used in this manner to assess the accuracy of numerical solutions and the segmentation used in the computer program. As a final check, sum all deficits calculated employing Equation 3-1 and/or

Table 3-18 formulations and compare the total deficit to that obtained from a comparable computer run with the computer doing the summation. The same reaeration equation and temperature adjustments should be employed in each computation.

The exercise suggested above will verify the validity and accuracy of a specific program code as set up in a particular computer installation. This activity will also provide valuable insights and experience for the individual who has not been involved in hands-on modeling of dissolved oxygen.

Identify Inputs: Loads, Rate Coefficients, Transport

At this point in a project, the equations and formulations that are to be used in the analysis have been selected. A computational technique has been chosen and placed on a computer facility. The computer program outputs for each of the reactions and loading types have been checked against the appropriate analytical solutions. The next step in the project is to unravel the cause and effect relationships that are controlling dissolved oxygen concentrations in the stream or river. The first step, model calibration, essentially consists of comparing dissolved oxygen, BOD, NH_3, NO_2, and NO_3 profiles calculated by the model to observed data.

General--Due to the critical importance that input parameter selection has in any modeling effort, the utmost care must be taken to use the best possible estimates for values. The reader should refer to the discussion of this subject in the early part of Section III-C, Dissolved Oxygen Analysis, in Appendix A.

Loads--The loads, sources and sinks in category I (i.e., those that are significant for the site-specific situation) should be defined from direct measurements corresponding to each set of receiving water quality data secured. Loads, sources, and sinks in category II (those that are considered to control background conditions) can be estimated from a combination of isolated measure-

TABLE 3-19. TYPICAL RANGES OF LOAD FOR SOURCES*

Source	Range	Supplemental References
Domestic and Industrial Point Sources	NPDES Permits Compliance Reports	--
Upstream Background Levels:		
Dissolved Oxygen Deficit	0.5 - 2.0 mg/l	7, 5, Use Storet
BOD$_5$	0.5 - 3.0 mg/l	30, 31, 32, Use Storet
NH$_3$	0.05 - .27 mg/l	30, 31, 32, Use Storet
NO$_3$	.07 - .37 mg/l	30, Use Storet
Organic N	.05 - .50 mg/l	30, 31, Use Storet
Urban Storm Runoff, BOD	20 mg/l	32, 29
Organic N	1.4 mg/l	32, 29
TN	3.1 mg/l	32, 29
Combined Sewer Overflow, BOD	115 mg/l	32, 29
Organic N	3.8 mg/l	32, 29
TN	9.1 mg/l	32, 29
Nonpoint Loadings, BOD$_5$	6 - 60 mg/l	29
TN	0.8 - 1.3 mg/l	33
Benthal Oxygen Demand, S (gm/m^2/day)	1 - 10	5, 10, 2
Average Gross Photosynthetic Oxygen production P$_a$ (gm/m^2/day)	.3 - 18	2, 5, 10
Assimilation Number, (gm O$_2$/hr/gm Chl a)	.7 - 4.5	2
Average Algal-Respiration R$_a$ (gm/m^2/day)	.025 (Chl a)	5

*These can be used for Category II Sources with confirmation by site-specific data.

ments, for confirmation, supplemented by information in the literature. Table 3-19 presents data on typical ranges of values for loads, sources, and sinks and lists references that may be employed with confirming measurements and local study results to develop estimates for background conditions for category II sources. Category I sources usually require direct site-specific measurements.

Point source inputs from domestic and industrial waste treatment plants will usually be in category I sources and should be measured during each period that receiving water quality data are collected in an allocation study. For historical data, compliance reports for NPDES permits can be used. Data on upstream contributions can often be obtained from Storet and 208 or other studies. Runoff loads from combined and separate sewer systems, if these are necessary to include as category II sources, can be estimated from Table 3-19, supplemented by a modest amount of local data from the allocation study or a recent 208 program study.

Reaction Coefficients. Reaction rate coefficients, in contrast to loads, sources, and sinks, generally cannot be directly measured under natural conditions. Indirect measurements, supplemented by calculations, provide one of the most reliable techniques for estimating reaction rates. In addition, laboratory and field experiments can provide information on the relative range of reaction rates. However, laboratory conditions usually are sufficiently different from natural conditions that direct application of experimentally derived reaction rates to natural streams is not valid. Illustrations of this difficulty are found in measurements of BOD oxidation and nitrification rates under laboratory conditions, which generally cannot be directly used to estimate these reaction rates in natural systems.

Assuming that inputs to the stream or river are adequately characterized by direct measurement, the basic objective of the model calibration process is to define a system of reaction rates that when combined with the measured inputs, yield calculated contaminant profiles that are consistent with observed data. The reaction rates employed in water quality modeling are based on physical, chemical, and biological principles or approximations to processes that can be examined under more or less controlled laboratory or field conditions. Therefore, two significant constraints must be imposed upon the values for reaction rates employed in water quality modeling calculations. These constraints are:

- All reaction rates must be uniform in space and in time unless variations are systematically related to identifiable system characteristics or processes.
- The rates and the formulations for systematic variation used in specific water quality modeling projects should fall within the range reported in the literature.

As an example, the first constraint allows variation of reaction rates as a function of temperature and factors such as stream depth or bed characteristics, light, nutrients, etc., but prohibits spatial variation of reaction rates to "fit the data" or changes in reaction rates used to characterize different sets of data or periods within a data set. This constraint is basically an outgrowth of the deterministic approach to water quality modeling that has classically been employed and is suggested for waste load allocation studies. The first constraint can be tested readily by application of the question: Do reaction rates vary in a systematic manner that can be identified and demonstrated to be phenomenologically realistic?

In one sense, the second constraint is easier to test since the range of reaction rates reported in the literature has been summarized (2). In a more fundamental sense, this constraint is really very difficult to evaluate. The objective of the modeling effort is to develop a quantitative assessment of factors controlling water quality. An evaluation of the portion of the water quality responses that is quantitatively undefined should be included in this assessment. These latter responses will usually be important in the decision-making process. The range of values reported in the literature for individual reaction rates is usually quite large. Thus, while a series of reaction rates may be technically within the numerical range reported in the literature, there may be a consistent bias in the values assigned that attempts to compensate for the effects of a phenomenon not fully incorporated in the analysis. This concern is the essence of the second constraint and is very difficult for both analyst and reviewer to identify.

The following segments of this report present tabulated ranges of reaction coefficients, formulations to calculate probable values, and discussions of methods for defining site specific reaction rate coefficients. It should be recognized by the analyst and included in the decision-making process that the level of available technical know-how will not provide a basis for _a priori_ assignment of site-specific reaction coefficients. This results from two factors.

The first factor is associated with an incomplete technical understanding for particular reactions, of the processes, pathways, and system characteristics that influence the speed of the reaction. Examples of reactions where incomplete tech-ical understanding is of importance are the nitrifition reactions and sediment oxygen utilization. For both of these reactions there are qualitative hypotheses available describing the pathways and probable influence of system characteristics, but quantitative _a priori_ prediction of reaction rates even in controlled laboratory experiments have not been made.

The second factor that limits the ability to assign site-specific reaction rates in waste load allocation projects is associated with the need to properly average reaction rates or system characteristics over segments of a natural stream or river. The process of atmospheric reaeration is an example where the second factor is probably of major significance. There is general agreement that liquid film resistance at the air-water interface determines the reaeration coefficient. Various formulations have been developed that attempt to relate system characteristics such as depth, velocity, rate of energy loss, etc. to the reaeration coefficient. The liquid film resistance is controlled by local system characteristics such as depth, velocity, rate of energy loss, etc., therefore, the various formulations really provide different approximations for the process of averaging local phenomena over a segment of a stream or river.

In view of the limitations in technical knowledge and the associated difficulties in properly averaging local processes over large segments of a stream or river, a priori assignment of reaction rate coefficients in waste load allocation projects carries considerable uncertainty. The process of model calibration and verification is the preferred method for narrowing the probable range of site-specific reaction rates. The ranges of coefficients presented in this section may also be reduced based upon the judgement of the analyst derived from calculations and data on the site-specific project or experience with other similar systems. In any event it is unlikely that a single set of coefficients can be developed; thus, there is the need to carry out a formal or informal sensitivity analysis. The range of probable responses from this sensitivity analysis should be carried into the decision-making process.

The range of reported rate coefficients employed in dissolved oxygen analysis of streams and rivers is summarized in Table 3-20. This table also tabulates the range of temperature formulations available. Table 3-21 presents a summary of the reaeration formulas available. A review of the supplemental references indicated in Tables 3-20 and 3-21 would be useful. These references provide discussions of the phenomena that control the values of the various rate coefficients, and tabulations of the specific rates found in various field studies, laboratory experiments, and modeling projects. Table 3-20 also presents some indication of the usual range of the various coefficients.

BOD Oxidation and/or Removal Rate (K_d/K_r). In most stream and river waste load allocation studies, the BOD oxidation rate K_d is of primary importance. The classical approach to evaluation of this rate is to obtain long-term stream carbonaceous BODs at various locations. A nitrification-inhibited test procedure is recommended. A semi-log plot of log BOD_u vs time of travel has a slope equal to $-K_d$, or in cases where settling, volatilization, or other non-oxygen using phenomena reduce BOD, the slope equals $-K_r$. This procedure has been modified by plotting log BOD_u vs distance with a slope equal to $-K_d/U$. K_d may be directly calculated from the slope and average velocity (U). Further modifications in the procedure for evaluation of K_d involve plots of the log of BOD_5 vs distance. The various approaches can yield slightly different values for the BOD oxidation rate. The values obtained by any of the methods are usually adequate for the analysis in view of the usual scatter in measured oxygen demand (BOD_5 or BOD ultimate) encountered. In cases where the variation in K_d found with the several methods is significant, the approach using long-term BOD vs time of travel is preferable.

TABLE 3-20. RANGE OF REACTION RATES FOR DISSOLVED OXYGEN ANALYSIS

Reaction Rates	Range	Usual Range	Supplemental References
Organic Nitrogen to NH_3 (1/Day)	.005 - .4	.05 - .3	31, 34, 35, 9
NH_3 to NO_2 (1/Day)	.003 - .5	.05 - .3	31, 34, 35, 9
NH_3 to NO_3 (1/Day)	.040 - 2.5	.05 - .3	31, 32, 34, 35, 9
NO_2 to NO_3 (1/Day)	.090 - 10.0	.10 - 1.0	31
Setting Rate (1/Day)	.001 - .1	.01 - .1	31
BOD Oxidation "K_d" (1/Day)	.020 - 3.4	0.10 - 1.0	31, 34, 9
BOD Removal "K_r" (1/Day)	.020 - 3.4	0.10 - 1.0	31, 34, 9
K_L Film Coefficient Reaeration (Ft/Day)	1.5 - 50	1 - 5	31, 34, 9
Reaeration Rate (1/Day) K_2 or K_a	.1 - 50	-	31, 34, 9 (use formulation)
BOD_u/BOD_5	1.1 - 4.0	-	31
Temperature correction factor θ			
NH_3 Oxidation	1.0548 - 1.0997	-	31, 34, 9
Benthic Uptake	1.041 - 1.09	-	31, 34
BOD Oxidation	1.02 - 1.09	-	31, 34, 9
Reaeration Rate	1.008 - 1.047	-	31, 34, 9

Notes: (1) Formulation normally is: $K_t = K_{20°C} * \theta^{(T-20)}$

(2) All rates to base "e".

TABLE 3-21. REAERATION COEFFICIENTS FOR STREAMS

Original Reference	Formulation for k_2 (base e)	Units of Variables	Development Conditions	Discussed By
O'Connor and Dobbins (1958)	$\frac{\left(D_M \cdot U\right)^{0.5}}{H^{1.5}}$	Any compatible set of units	For streams displaying isotropic turbulence. The observed data had the following characteristics: $1' < H < 30'$; $0.5 < U < 1.6$ fps; $0.05 < k_a < 12.2$/day.	Covar (1976) Holley (1975)
Churchill, *et al.* (1962)	$\frac{11.6U^{0.969}}{H^{1.673}}$	U-fps H-feet k_2-1/day	Based on observed reaeration rates below dams from which oxygen deficient water was released. $2' < H < 11'$; $1.8 < U < 5$ fps.	Covar (1976) Holley (1975)
Owens, *et al.* (1964)	$\frac{21.7U^{0.67}}{H^{1.85}}$	U-fps H-feet k_2-1/day	Oxygen recovery monitored for six streams in England following deoxygenation with sodium sulfite. $0.1 < U < 5$ fps $0.4' < H < 11'$	Covar (1976)
Langbein and Durum (1967)	$\frac{7.6U}{H^{1.33}}$	U-fps H-feet k_2-1/day	Based on synthesis of data from O'Connor and Dobbins (1958), Churchill, et al. (1962), Krenkel and Orlob (1963), and Streeter, *et al.* (1936).	Covar (1976)
Isaacs and Gaudy (1968)	$\frac{7.03U}{H^{1.5}}$	U-fps H-feet k_2-1/day	Developed using regression analysis on data collected from a circular trough with recirculating water.	Covar (1976)
Negulescu and Rojanski (1969)	$4.74\left(\frac{U}{H}\right)^{0.85}$	U-fps H-feet k_2-1/day	Developed from a recirculating flume with depths less than 0.5 feet.	Covar (1976) Elmi (1975)
Tsivoglou (1967, 1972) Tsivoglou and Neal (1976)	$0.054\left(\frac{\Delta h}{t}\right)$ @25°C	Δh-feet t-hours k_2-1/hour	Gas tracer technique used. $25 < Q < 3000$ cfs	Covar (1976) Elmi (1975)
Foree (1976)	$0.30 + 0.19S^{1.2}$ @ 25°C	S-feet/mile k_2-1/day	Gas tracer technique used on small streams in Kentucky. $0 < S$ 42 feet/mile $.02 < q < 2.02$ cfs/sq.mile	Foree (1976)
Gloyna, *et al.* (1971)	$6.86\,\frac{U^{0.703}}{H^{1.05}}$	U-fps H-feet k_2-1/day	Regression analysis used on data for which $9.8 < k_2 < 28.8$/day	Gloyna, *et al.* (1971)

(continued)

Table 3-21. REAERATION COEFFICIENTS FOR STREAMS (concluded).

Original Reference	Formulation for k_2 (base e)	Units of Variables	Development Conditions	Discussed By
Thackston and Krenkel (1969)	$0.000299\left[1+\left(\frac{U_*}{\sqrt{gH}}\right)^{1/2}\right]\sqrt{\frac{Sg}{H}}$	--	--	Thackston and Krenkel (1969)
O'Connor and Dobbins (1956)	$\left(D_M\frac{U}{H}\right)^{1/2}$	--	Valid for low velocities and anisotropic conditions.	Hydrologic Engineering Center (1974)
Krenkel and Orlob (1962)	$9.9\cdot10^{-5}\ D_L^{1.15}0_H^{-1.95}$	D_L-ft^2/min k_2-1/min H-ft	Based on artificial channel observations.	Elmi (1975)

Notes:

1. All expressions given to base e and for 20°C unless otherwise noted
2. Definition of symbols:

D_M = molecular diffusion coefficient ($8.1\cdot10^{-5}$ ft/hr @ 20°C) for oxygen in water
U = stream velocity
H = average stream depth
Δh = change in stream elevation between two points
t = travel time between two points for which Δh measured
S = slope (change in water surface elevation divided by distance)
q = specific discharge (cfs per square mile of drainage area)
D_L = longitudinal dispersion coefficient
U_* = bed shear velocity

In situations where water quality is dominated by waste discharges from well-operated secondary treatment plants, the rate of BOD removal K_r is usually equal to the rate of BOD oxidation K_d. For other instances, settling, volitalization, and other processes may remove BOD without directly utilizing the oxygen resources of the system. As an illustration, urban runoff loads from separate and combined sewer systems usually have significant organic loads associated with solids. If these solids settle in the stream or river, all of the observed BOD removed does not use oxygen immediately. The organics that settle contribute to the oxygen demand of the bottom, which may be exerted over a time frome of months to years. Caution should be exercised when attributing BOD removal to settling when the discharge is from a biological treatment system, in that TSS (Total Settleable Solids) in such discharges often remain in suspension. This process can be observed in the semi-log plots indicated previously. The slope of the plot is initially large and then reduces as shown in Figure 3-3. Normally, the large slope in the first part of the curve is employed to define K_r, the BOD removal rate, while the second lesser slope is assumed to represent K_d, the deoxygenation rate. As shown on the figure, $K_r > K_d$ for the first reach and $K_r = K_d$ in the second reach. In these situations, observations of the stream bottom should be made to confirm that settling is responsible for the higher removals observed in reach one. This confirmation is necessary as other phenomena may produce similar BOD profiles. Comparable profiles could be observed in streams where, for example, reach one was characterized by a shallow rocky bottom stream and reach two was deeper with a silt bottom. In such a case; $K_r = K_d$ in both reaches; the stream geomorphology produces a higher deoxygenation rate in the first reach. The lower portion of Figure 3-3

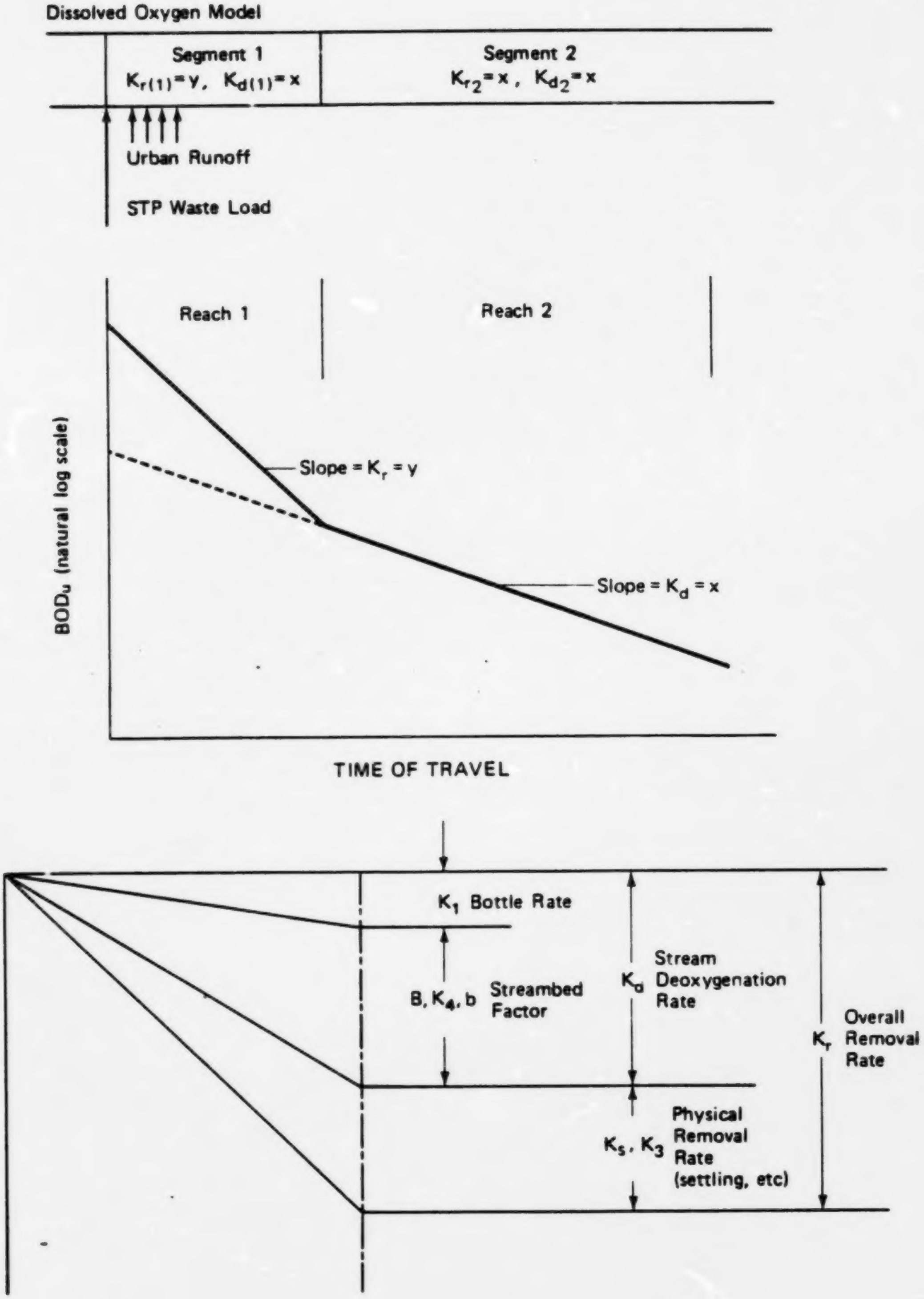

Dissolved Oxygen Model
Segment 1
$K_{r(1)} = y$, $K_{d(1)} = x$
Segment 2
$K_{r_2} = x$, $K_{d_2} = x$
Urban Runoff
STP Waste Load
Reach 1
Reach 2
BOD_u (natural log scale)
Slope = K_r = y
Slope = K_d = x
TIME OF TRAVEL
K_1 Bottle Rate
B, K_4, b Streambed Factor
K_d Stream Deoxygenation Rate
K_r Overall Removal Rate
K_s, K_3 Physical Removal Rate (settling, etc)

Figure 3-3. Typical BOD removal curves with settling or other non-oxygen using processes of BOD removal.

illustrates schematically the relationship between various components of the BOD removal rate.

A wide range of values for the carboneaceous (CBOD) deoxygenation rate (K_d) has been observed in natural water systems. Two fundamental factors are thought to influence, or determine the magnitude of this rate in rivers and streams. These are:

Stream Geometry and Geomorphology

Other things being equal (temperature, reactivity of the organic compounds comprising the CBOD), the observed net overall deoxygenation rate along a stream reach should be related to the number of microorganisms that come into contact with the CBOD constituents. These organisms will be present both in the water column and on the stream bed as attached organisms. Under any set of conditions of temperature and reactivity, the relative amounts of "attached" bacteria on the stream bed represent the biggest variable from one stream system (or reach) to another.

Their relative numbers can be considered to be related to:

- The area of streambed, which can be defined by the wetted perimeter. However, the "effective" relative numbers might be thought to include some expression of the wetted perimeter in relation to the total volume of water contained in the stream section--this to reflect the opportunity for contact.
- The cross-sectional area divided by the wetted perimeter is designated the hydraulic radius (R_h)--and is closely approximated by depth for roughly rectangular streams in which the width is large compared with depth. For example, where width/depth ratio is 20 or more, R_h is approximated by depth within 5 to 10%.
- The nature of the stream bed, since this can influence the extent to which a bacterial population can establish itself, and the stability

> of the population once initially established. Stable rocky stream beds are considered to provide more favorable conditions for maintaining a large, effective population--than are unstable, sandy or silty channels where hydrologic perturbations are prone to smother or scour out populations that gain a foothold.

Based on these considerations, the deoxygenation rate (K_d) could be expected to increase with decreases in stream depth (or the relationships for which depth is a surrogate) and to increase with the stability of the stream bed.

Reactivity of Organic Compounds Comprising CBOD

Raw sewage and primary effluent will have a higher proportion of easily bio-oxidized constituents than would a secondary effluent, which has been exposed to intensive biological contact. Similarly, application of AWT to a wastewater discharge can be expected to remove all but the more refractory organic compounds--so that what remains for discharge to the stream consists of organic compounds that will be biooxidized more slowly than those discharged where lower levels of treatment are applied.

The reactivity of organic compounds from industrial waste discharges comprising CBOD would depend on both the level of treatment applied and the type of industry involved. For example, treated pulp mill effluents decay at very slow rates, but have high ultimate $CBOD_u/CBOD_5$ ratios. Although different waste discharges exhibit individual decay rates it is often assumed the decay rate for the river segment can be represented by a single decay rate.

Considering the number of factors that affect the rate of oxygen utilization in natural streams, all of which are incorporated in the coefficients, K_d and K_n, it is not surprising that a large range of values has been reported. It is also apparent why stream surveys are recommended to quantify site-specific values for these coefficients, in spite of the expense, time and difficulty of such projects.

However, the occasion frequently arises when the available resources of time, personnel and money are inadequate to conduct such a survey and the question of assigning an appropriate stream coefficient is posed. Even in a case where a survey is conducted, and the stream coefficient evaluated, it will only be representative of the flow, loading and temperature at the time of the survey. The problem then arises of projecting a value that takes into account the effect of a different condition (higher treatment levels, drought flow conditions). The difficulty is further compounded when a development--municipal, industrial or agricultural--is proposed for a river site on which there is no or little data. In all these cases, the question of assigning an appropriate coefficient arises.

Figure 3-4 summarizes a carefully screened set of data on stream K_d values that have been established with high levels of confidence (via well-verified models).

A definite correlation between K_d and depth is evident in the range of stream depths between 1 foot and 5 or 6 feet. No data is available at stream depths less than one foot, and no clear relationship between K_d and depth is indicated for depths greater than 5 or 6 feet. Further, the range in potential values for K_d at any particular depth is quite broad--with about 6 fold differences between high and low ends of the range being observed.

A suggested relationship is shown by the band placed on the data in this figure. This is based on an interpretation based on a somewhat crude, but rational set of empirical rules. The rules have in turn been based on observations of stream system behavior in terms of K_d values made by a number

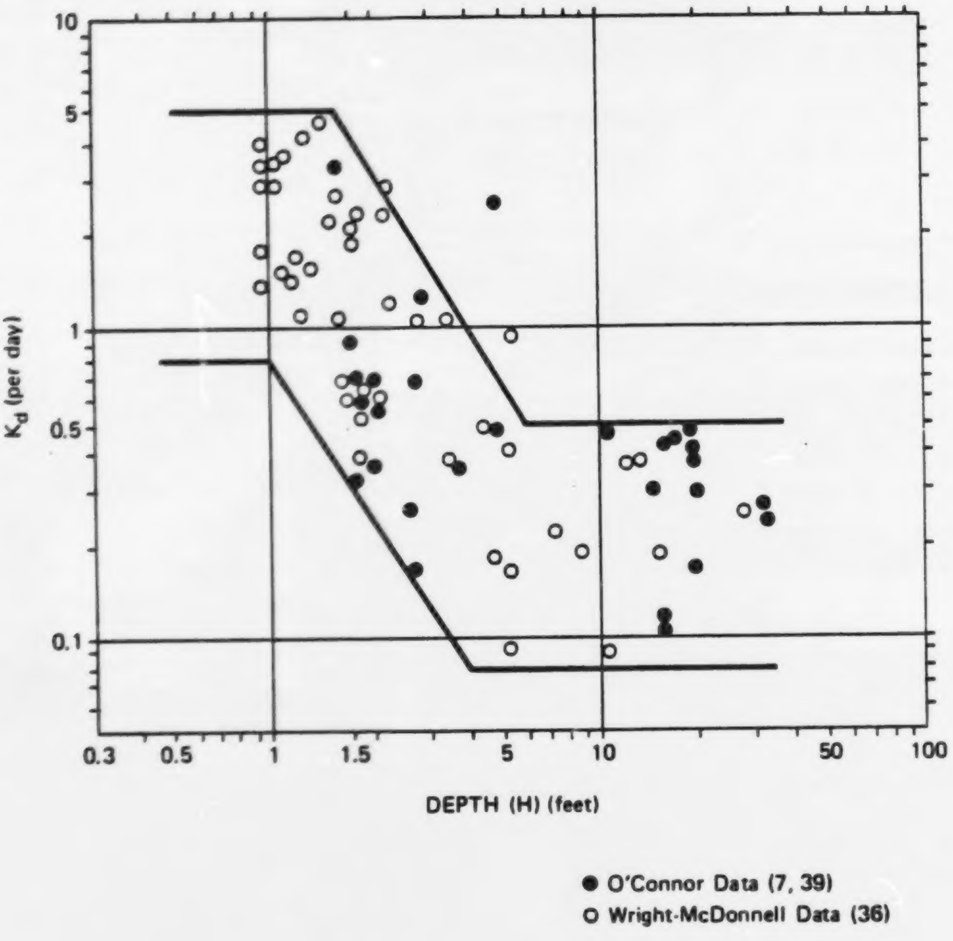


Figure 3-4. Deoxygenation coefficient (K_d) as a function of depth.

of investigators, and an integration of approaches that attempt to account for the independent effects of streambed characteristics, and level of treatment reflected in the laboratory bottle rate (K_1).

In the suggested relationship, K_d is shown to be dependent on average stream depth--over the range of stream depths between 1.0 to 1.5 feet and 5 or 6 feet. For depths less than 1 foot, the value of K_d is made independent of depth, reflecting the expectation suggested by various investigators that some limiting value will be reached. In this depth range, the value of K_d within the range shown will be determined by streambed characteristics, as described later.

For stream depths greater than about 5 or 6 feet, K_d is also made independent of stream depth. In this range the principal determinant of K_d is the relative biodegradability of the waste, determined for municipal wastes by the level of treatment applied, and reflected by the laboratory derived reaction rate (K_1). Values for K_1 can be determined from long-term BOD tests.

In the 1 to 6 foot depth range, a combination of waste reaction rate (K_1), and benthic microorganism population, reflected by depth and streambed characteristics, is important.

O'Connor has proposed an approach (5) that provides a basis for narrowing the broad range of potential values, after estimating the order of the K_d value from Figure 3-4. His approach provides a basis for making a rational estimate based on coefficients derived from laboratory tests of the waste, and background receiving water conditions, such that the stream coefficients will reflect both of these effects. The framework is similar to procedures that have been proposed in the past and like them lacks the body of data required to independently evaluate the coefficients it incorporates. It is accordingly not presented as a quantitative procedure--but rather as a qualitative framework to aid the analyst's judgment in assigning values for K_d in specific situations.

The approach relates stream deoxygenation rates (K_d) to (1) the turbulence of the flowing water, (2) the physical characteristics of the streambed with respect to the volume of the water in contact with it, and (3) laboratory measurements of the BOD oxidation rate (K_1) from BOD bottle tests. It is emphasized again that the approach is basically qualitative. However, some of the elements can be determined; long-term BOD tests can be performed to establish a value or range for K_1, and reconnaissance can determine the general characteristics of the streambed (e.g., rocky, stable, intermediate or unstable). Furthermore, estimates of average depth can be made together with calculations to estimate changes in depth with stream flow. At worst--with no site-specific information on K_d under any conditions--such efforts will assist in narrowing the probable range for sensitivity analyses. This will probably prove to be important in many cases because the range indicated by Figure 3-4 may prove to be too broad for practical utility in performing a waste load allocation analysis. In cases where a site-specific evaluation of K_d has been made under some condition, estimates for other conditions will be significantly enhanced, because they will reflect the net effect at that site of the streambed factor.

The suggested integration of the three indicated effects is as follows. The mass rate of reaction, K_dLV, may be described in terms of the two components, the planktonic microorganisms dispersed in the volume of flowing water (V), and the attached benthic microorganisms on the bed surface (A).

Thus:

$$K_dLV = aK_1LV + bLA \tag{3-2}$$

in which

K_d = overall stream deoxygenation coefficient, 1/day
K_1 = BOD bottle reaction coefficient, 1/day
L = BOD concentration, mg/l
a = mixing factor
b = streambed factor

and dividing through by LV, where $H = \frac{V}{A}$

$$K_d = aK_1 + \frac{b}{H} \tag{3-3}$$

Equation (3-3) is represented as a rather crude, but rational, empiricism that can be used qualitatively to assist in estimating stream deoxygenation coefficients on the basis of bottle rates and the substantial effects of stream depth and bed conditions.

It is not presented as a quantitative mathematical expression for direct calculation of K_d values. A rigorous analysis of an adequate data set would be required to assign values to the coefficients a and b. It is not clear whether an adequate data base, which includes information on all pertinent parameters, exists. It has not been assembled to date.

However, while precise values for the above coefficients cannot be assigned, their order can be inferred from Figure 3-4. If the assumption of an upper limit to K_d in streams less than 1 to 1.5 feet deep, and dominated by bed characteristics, is correct, then values for the bed factor (b) will range between about 0.8 and 5.0. The value is projected to increase with increasing stability of the bed: O'Connor has suggested a range for (a) of 1.1 to 1.5, increasing with increasing stream turbulence.

The impact of BOD removal and/or oxidation rates that vary between sources can have a substantial influence on waste load allocations and decision making. The sensitivity of each individual system will be related to the ultimate BOD load associated with the different sources and the flow, system geometry, etc. Table 3-22 contains results of calculations illustrating the influence of BOD oxidation rates that vary with type of loading. The variations in reaction rates used are within the range that could be anticipated. The average stream reaction rates calculated by the two techniques indicated are reasonably close. The average stream K_d is 60% larger than that for load A and 80% of the load B value (Row 1). Removal of portions of load A would result in an increase in the average stream K_d (Row 3), while treatment of load B results in a decrease in the average stream K_d (Row 2). These calculations employ the basic first-order BOD equations and do not consider the further complication that K_d may change as a result of the treatment process. The basic procedures to be discussed for model calibration and verification will provide a framework for defining the significance of the differences in reaction rates and a method for estimating the rates for individual sources if necessary.

Nitrogen Reaction Rates (K_n)

The rate of conversion of organic nitrogen to ammonia may be calculated in a fashion similar to that employed to define BOD removal and oxidation

TABLE 3-22. ILLUSTRATION OF THE INFLUENCE OF BOD REACTION RATES WHICH VARY BY LOADING TYPE ON THE CALCULATED AVERAGE STREAM "K_d"

Row	Load Type	UOD mg/1	K_d 1/day	Load Type	UOD	K_d 1/Day	Average BOD_5 vs Time 1/Day	Stream K_d UBOD vs Time 1/Day
(1)	A	5.0	.1	B	12	.2	.175	.16
(2)	A	5.0	.1	B(1)	3	.2	.140	.13
(3)	A(2)	2.5	.1	B	12	.2	.180	.17

Notes: (1) 75% removal of Source B; no change in reaction rate

(2) 50% removal of Source A; no change in reaction rate

rates. In this instance, the hydrolizable organic nitrogen fraction should be employed. If significant quantities of organic nitrogen are hydrolized to ammonia, then the ammonia oxidation rate can be obtained only by trial and error calculations comparing calculated and observed profiles of organic nitrogen, ammonia, nitrite, and/or nitrate. The comparison of observed and calculated profiles for the oxidized nitrogen forms is a key element in determining and confirming nitrification rates. If nitrate is not formed, considering appropriate mass balances, the presence and importance of nitrification has not been demonstrated. Speculation on nitrate removal mechanisms, which almost balance ammonia oxidation uniformly in space and time, such as bottom denitrification or plankton uptake, should not be considered as adequate justification for decisions in waste load allocations projects.

In systems where hydrolysis of organic nitrogen is not significant, the nitrification rate may be approximated using a semi-log plot of ammonia vs time of travel. Again, the oxidizable forms of nitrogen are a key element in demonstrating the existence and importance of nitrification.

Projections of nitrification under future conditions can be characterized by two basic questions:

- Will nitrification occur?
- If nitrification occurs, what is the rate at which it will occur?

For situations where nitrification is presently occurring, it may be assumed that it will continue to be present in the future. This assumption should be used if existing nitrification is confined to the downstream oxygen recovery zones and present critical dissolved oxygen levels are below approximately 2-3 mg/l, which can inhibit nitrification. Normally, nitrification is assumed to occur under future conditions at the maximum observed rates. The "normal" assumptions indicated above should be reviewed in the context of the individual stream. As an illustration, current theory asserts that nitrifying bacteria may compete most effectively if they can find sites on particles. Therefore, major differences in suspended solids levels, the ratio of wetted area to stream cross-section or the nature of the stream bed (rocky or muddy), between reaches should prompt reevaluation of these "normal" assumptions.

For situations where nitrification is not observed under existing conditions, projections should be made with and without nitrification under future conditions. The processes controlling nitrification are not sufficiently understood so as to permit definitive projection of the presence or rate of nitrification in the future if it is not currently observed. The nitrification projections can be employed in a staged approach to construction of facilities.

Reaeration Coefficient K_a

Procedures are available for estimating the reaeration coefficient. Table 3-21 presents a summary of the formulations that have been developed. In addition, existing field procedures, such as tracers, can be employed to directly measure the reaeration coefficient. For each major project, field measurements should be obtained for at least one flow condition over the length of stream to be studied. A distinction needs to be made between the technique for measurement of K_a and the equations used to project K_a for other conditions of flow, depth, and temperature. Use of measurement techniques within a project can be combined with several of the reaeration formulas to make projections to other conditions.

Sediment Oxygen Demand (SOD)

Bottom sediments of all rivers and streams utilize dissolved oxygen from the water column. The range of reported values is presented in Table 3-19. The oxygen utilized by sediment is associated with decomposition of organic matter and may also be influenced by oxidation-reduction reactions that occur in the interstitial waters of deep layers of sediment. Sediments that are low in organic content can utilize dissolved oxygen at a rate of on the order of 1 gm/m^2-day, or lower, while sediments with moderate organic content can have an oxygen utilization rate that is from three to six times this value. Sediment oxygen demand values above 6 gm/m^2-day are usually associated with high organic sediments that have continual additions of new organic matter. In general, all waste load allocation projects should consider inclusion of a factor for bottom oxygen utilization. This may be estimated from visual observation of the sediment, or by direct laboratory or field measurement of the SOD. Field measurements employing sediment capping and oxygen utilization data are preferred.

The importance of SOD in a site specific situation may be estimated by employing the following equation:

$$\text{Maximum SOD Deficit} = \frac{S}{H\,K_a}$$

where: S = SOD (gm/m^2-day)

H = depth (m)

K_a = reaeration coefficient (1/day)

For systems with large velocities and high rates of oxidation, the maximum SOD dissolved deficit may not be reached in the critical region (x_c). The SOD at this location may be calculated by:

Deficit from SOD at $x_c = \left[\frac{S}{H\,K_a}\right] \left[1 - e^{\frac{-K_a}{U} x_c}\right]$

SOD effects should be considered in all waste load allocation studies, and included in the calculations where these effects will be significant. Omission tends to result in an overestimate of the benefits from point source treatment.

Photosynthetic Activity

Plankton, periphyton, or rooted aquatic plants can cause significant fluctuations in dissolved oxygen concentrations, over the course of a day, when they are present in a stream reach in large enough concentrations. Where the magnitude of the diurnal variation is appreciable, violations of DO standards can occur for part of each day, even though the average daily concentration may be acceptable. In such cases, the effect of photosynthetic activity must be considered in performing a waste load allocation. The user is referred to Chapter 2 - "Nutrient/Eutrophication Impacts" for rivers and streams for guidance on procedures for incorporating photosynthetic effects into a waste load allocation study.

Transport. Transport can usually be defined by a series of measurements. There is a need to develop a reasonable flow balance with, if possible, some internal checks for consistency. USGS gaging records are sufficiently accurate for dissolved oxygen analysis and waste load allocations. It may be necessary to supplement existing gages with flow measurements and bottom cross-section profile surveys at additional locations. Furthermore, all major studies should conduct a time of passage study using dye or other tracers. These studies provide data to confirm flow balances, define stream velocities by reach, define the amount of longitudinal dispersion present, and indicate lateral mixing effects to aid in locating sampling points.

Structuring a Site-Specific Water Quality Model

The essential procedures for developing a site-specific water quality model consist of a combination of data collection and analysis. A model is used with the results of these activities to provide the information needed for projections, waste load allocations, and decisions. The steps in the process are identified in Figure 3-5. Before discussing each step, it is advantageous to present some definitions and requirements. Both model calibration and verification consist of comparisons of observed and calculated water quality profiles. These comparisons are defined to include:

- Comparisons of all significant water quality variables at the same time for each data set.
- Consistent loads, rate coefficients, and transport systems should be used for each data set and for all segments of a data set.
- Loads, sources, sinks, reaction rates, and transport should be constant in time and space unless systematic variations can be associated with definable processes or direct measurements (such as flow, temperature, etc).
- Comparisons of calculations and observations for two or more dissimilar conditions are required.
- The comparisons must be carried out on the same space and time scale that will be used for projects.

To clarify this latter point, steady-state, quasi steady-state and time-variable models must be compared to appropriate data. As an example, time-variable models should be calibrated and verified against time-variable data. This means that data at t=0 are used for initial conditions in the model, and

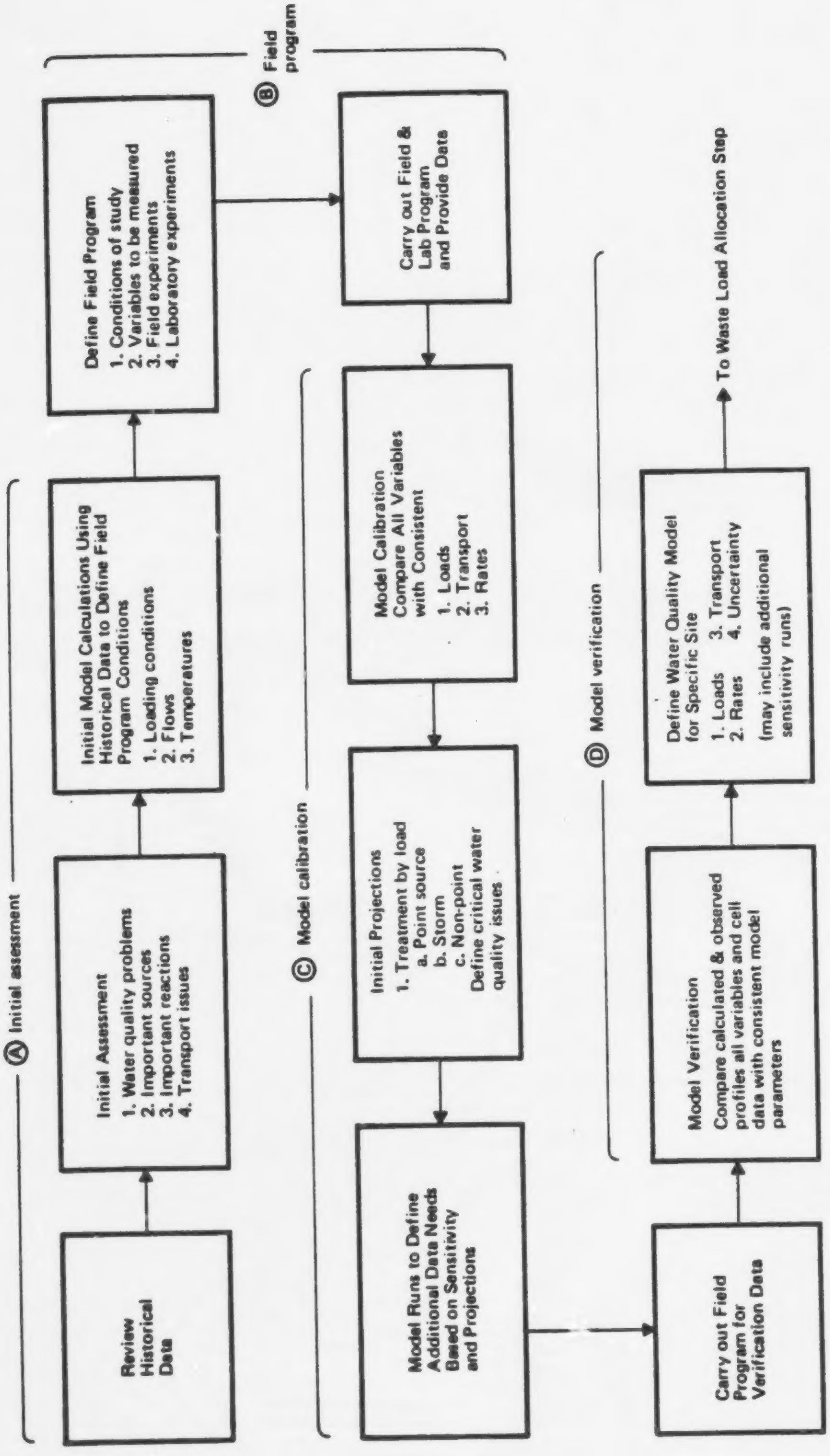

Ⓐ Initial assessment
Review Historical Data
Initial Assessment
1. Water quality problems
2. Important sources
3. Important reactions
4. Transport issues
Initial Model Calculations Using Historical Data to Define Field Program Conditions
1. Loading conditions
2. Flows
3. Temperatures
Define Field Program
1. Conditions of study
2. Variables to be measured
3. Field experiments
4. Laboratory experiments
Ⓑ Field program
Carry out Field & Lab Program and Provide Data
Ⓒ Model calibration
Model Runs to Define Additional Data Needs Based on Sensitivity and Projections
Initial Projections
1. Treatment by load
a. Point source
b. Storm
c. Non-point
Define critical water quality issues
Model Calibration
Compare All Variables with Consistent
1. Loads
2. Transport
3. Rates
Ⓓ Model verification
Carry out Field Program for Verification Data
Model Verification
Compare calculated & observed profiles all variables and cell data with consistent model parameters
Define Water Quality Model for Specific Site
1. Loads 3. Transport
2. Rates 4. Uncertainty
(may include additional sensitivity runs)
To Waste Load Allocation Step

Figure 3-5. Steps in development of site-specific water quality model.

comparisons of calculations and observations are made at $t=t_1$, $t=t_2$, $t=t_3$ ... $t=t_n$, as appropriate. A computer program that solves the stream dissolved oxygen equations by numerical integration in time should not be considered as a verified or calibrated time-variable model if steady-state calculations are used. A time-variable, site-specific model is defined as one that has been compared to time-variable data.

Initial Assessment

The first three steps in Figure 3-5 contribute to the initial assessment activity. The historical data are reviewed and employed in conjunction with initial model runs, which compare calculated and observed water quality to:

- confirm existing or future water quality problems.
- define the loads, sources, and sinks that control water quality.
- define the important reactions that control water quality.
- define issues in the area of stream transport that must be resolved.

This initial assessment is the first step of the process aimed at understanding the factors controlling water quality. Sources are placed in the appropriate category. The specific output from this activity is a defined field program of data collection that specifically identifies and defines:

- what sources will be measured
- when and at what frequency source measurements will be obtained
- under what conditions of load, temperature, and stream flow will data be obtained.
- under what seasonal flow regimes transport studies should be performed.

- what special studies are required and under what conditions
  - sediment oxygen demand surveys
  - reaeration evaluation
  - long-term BOD studies
  - nitrifier organism counts
  - light and dark bottle P-R studies
  - diurnal oxygen measurements
  - others

Each site-specific problem will require an individual analysis. As an illustration, consider the following example:

Analysis of historical data using a model indicates that the important category I sources are: 1) point loads, 2) upstream BOD, and 3) sediment oxygen demand. Calculations using the model are shown in Figure 3-6. These suggest that in addition to the usual summer low-flow survey, a higher-flow survey would provide data that could differentiate the effect of the upstream BOD from the point source loads. Furthermore, sediment oxygen demand measurements will be required. In addition, a fall-winter lower temperature survey will provide data that differentiate the effects of the sediment oxygen demand from the upstream and point source BODs. This can be done because the temperature effect and spatial distributions of dissolved oxygen deficit are different for sediment oxygen demand and BOD. In addition, the calculations suggest that the system dissolved oxygen is sensitive to the reaeration rate; therefore, reaeration experiments with time of travel are required.

The initial assessment activity is a first full step in understanding quantitatively the factors controlling water quality. It is not a preliminary analysis; instead, the initial understanding is translated into a field and experimental program whose data output begins to challenge and strengthen the understanding of the system.

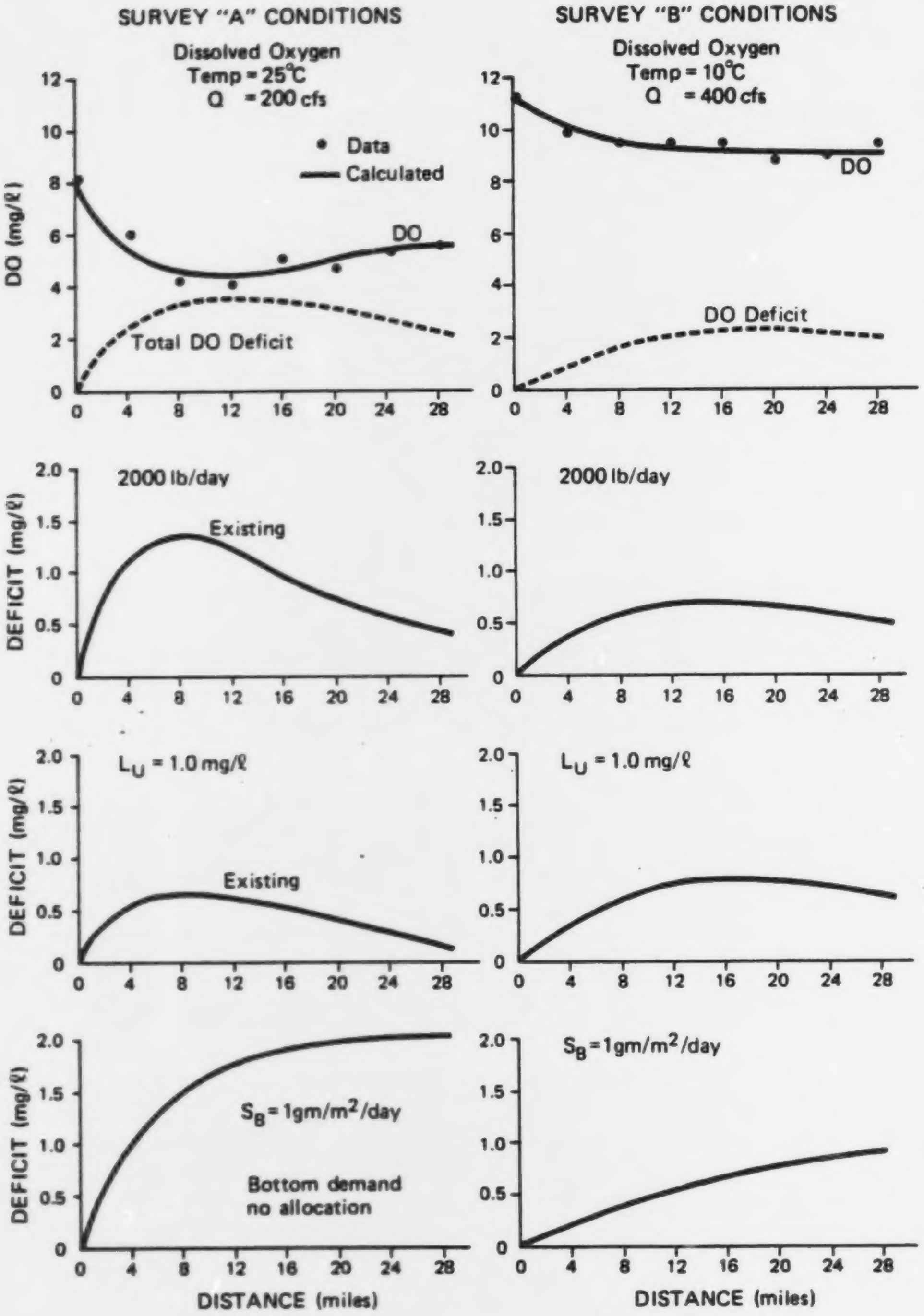


Figure 3-6. Illustration of the use of calculation to define survey periods.

Field Program. This task translates the results of the initial assessment into a practical field program that can be carried out on the river and in the laboratory using the resources and manpower required and/or available. The output from these activities is the required data on:

- loads, sources and sinks of dissolved oxygen
- rate studies (deoxygenation and reaeration)
- transport studies
- water quality

More detailed discussions are presented in Section 4.2.

Model Calibration. In these activities, the data from the field program are employed to define category I sources, reaction rates, and transport. Water quality calculations using the model are developed for the conditions of loading, flow, and temperature associated with each of the water quality data sets. These conditions include those associated with the historical data and the data collected in the waste load allocation study. Adjustments in the value of reaction coefficients and category II loads must be made in a consistent fashion for all conditions. The results of these activities are a set of consistent model parameters, which are then employed to develop water quality calculations for the conditions associated with all available data sets. Comparisons of calculated and observed water quality profiles should be developed. The model runs and calculations employed to search for and define the series of consistent coefficients should be retained since they can provide an indication of system sensitivity.

At this stage in the modeling process, a calibrated model has been developed. The next step involves a test of the adequacy of the model in terms of decisions required in the waste load allocation study.

Model Adequacy. Recent research activities (37) in modeling eutrophication in lakes have begun to explore the use of simple statistical comparisons in an attempt to quantify model adequacy. These techniques could be a supplement to the qualitative comparisons of observed and calculated water quality profiles. Three techniques that have been used are

- comparison of means
- regression analysis
- relative error

Comparison of Means. The mean of the observed data is compared to the mean of the computed profile for the comparable conditions of loading, transport, and temperature. The Student's t-probability density function is employed for the comparison of the means.

Regression Analysis. Calculated concentrations and observed data are considered as paired points in the test equation:

$$X = \alpha + \beta C + E \qquad (3\text{-}4)$$

α and β are the true intercept and slope, respectively, between the calculated values C and the observed data X. E is the error of X. The regression analysis assumes that calculated value C is known with certainty and that the error E is in the measured data X. Of course this is not necessarily a realistic assumption. Standard linear regression statistics can compute the square of the correlation coefficient r^2 (% variance accounted for) and the standard error of estimate representing the residual error between data and model. Estimates of the slope and intercept may be obtained and a test of significance developed.

Relative Error. Calculated concentrations and observed data are considered as paired data and the relative error is calculated by:

$$e = \left| \frac{X - C}{X} \right| \qquad (3-5)$$

The relative error may be aggregated across time or space, and the cumulative frequency of error can be computed. Estimates can be made of the median relative error as well as the 10% and 90% frequency of error. This statistic is poorly behaved at the upper tail and at low values of X. The median error can be easily understood; therefore, if statistical representations of model adequacy are to be employed in a waste load allocation study, this is the suggested measure.

Statistical measures of adequacy are in the early stages of research and should be employed recognizing that they provide, at the very best, a lower bound on the magnitude of the error.

It is imperative that the adequacy of the model be tested in the context of the loadings and reactions controlling water quality for each of the conditions that observed data are available. This can be accomplished by plotting the unit responses from each category I load for each condition. Qualitatively, the adequacy of the model is associated with the difference in magnitude of the water quality response to each source and the relative impact of each source under the several conditions examined.

As an example, Figure 3-7 contains the results of calculated dissolved oxygen and dissolved oxygen deficit profiles for two survey conditions. The first observation is that the dissolved oxygen profiles are different for each survey and, more importantly, the dissolved oxygen deficit profiles differ in shape and magnitude. Furthermore, the calculated influence of the point load discharge (2000 lbs/day) is almost twice as large during the

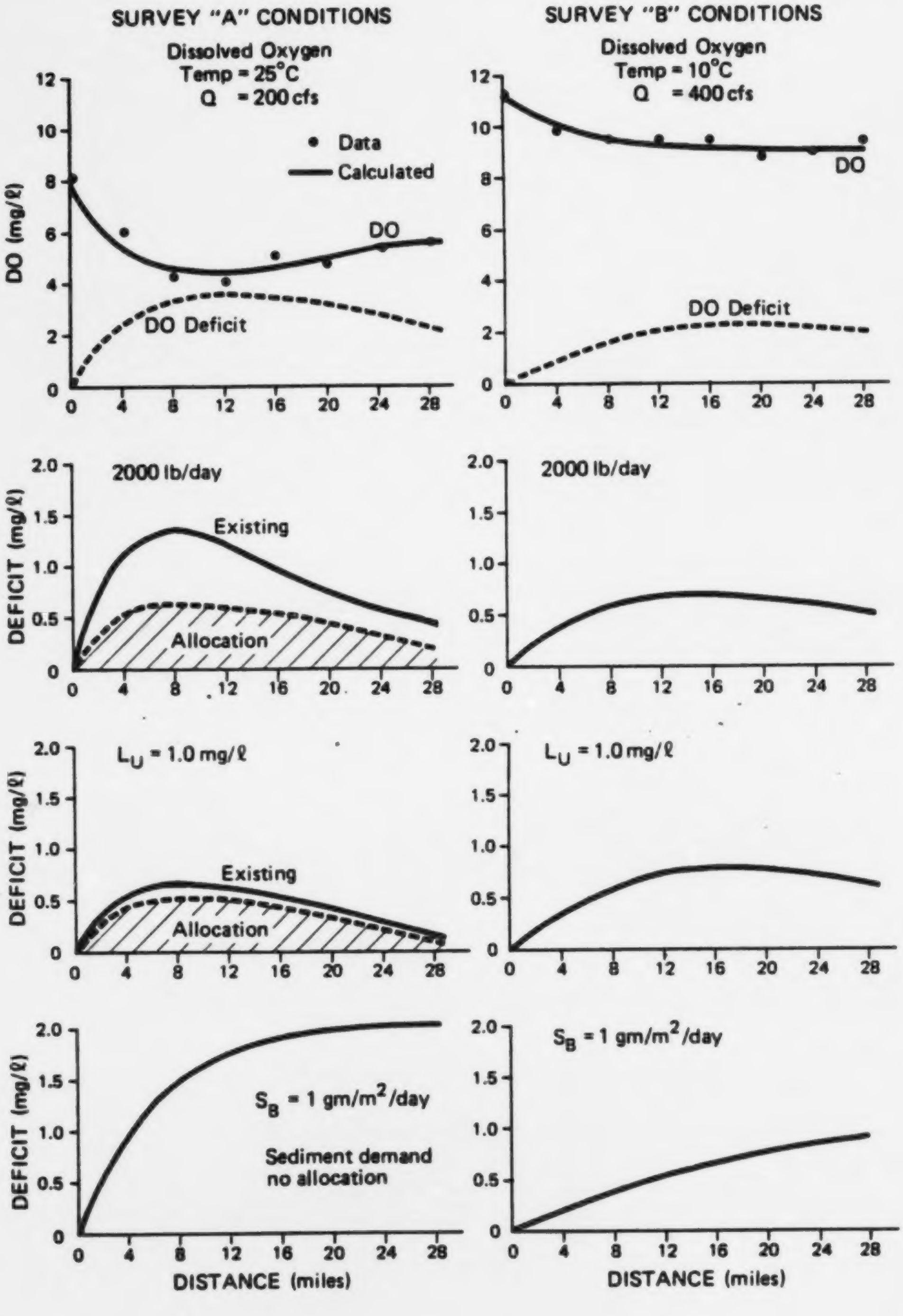


Figure 3-7. Unit responses at two conditions.

TABLE 3-23. ILLUSTRATIVE CALCULATIONS TO TEST MODEL ADEQUACY FOR ALLOCATIONS

Distance	DB(3,5)	D_{1000}(1)	D_L =.75(2)	D(o-c)(4)	DEF	DO
0	0	0	0	—	0	8.2
4	.98	.55	.45	.4	2.4	5.9
8	1.50	.67	.50	.4	3.1	5.2
12	1.77	.60	.45	.4	3.2	5.0
16	1.92	.50	.37	.4	3.2	5.1
20	2.00	.37	.28	.4	3.1	5.2
24	2.00	.27	.20	.4	2.9	5.4
28	2.10	.20	.10	.4	2.8	5.4

Notes: (1) 50% removal at point source

(2) 25% removal at upstream sources

(3) No removal

(4) D(o-c), measures of the variation between calculated and observed profiles

(5) DB: deficit caused by sediment oxygen demand.

understanding of system responses and testing the resulting level of knowledge. Variation or modifications of the approach will be necessary on a site-specific basis. These should be encouraged while retaining the essential objectives.

3.4 ASSESSING THE ADEQUACY OF MODEL VERIFICATION

In many cases, the procedures discussed in the previous subsection will indicate the need for obtaining additional water quality data sets under various conditions of loading, flow, and temperature. This will lead to data collection activities after the model calibration process, as indicated on Figure 3-5. This data collection should not be a random effort but should be directed toward obtaining information under different conditions to increase understanding of water quality responses and lead to the model verification activities.

The adequacy of a modeling effort in terms of decisions for waste load allocations is determined in part by the differences between calculated and observed data, as measured qualitatively by a visual evaluation of graphic representations in space and time. Statistical parameters, such as the median relative error, may also be used. Further measures of model adequacy are the diversity of conditions for which comparisons of data and calculations can be made and the ability to make waste allocation decisions. A final consideration is the sensitivity of the calculated water quality profiles to variations of reaction coefficients.

If calculated profiles are significantly changed by small variations in the values of reaction coefficients, several additional steps are necessary to evaluate model adequacy. The value of the coefficient to which the calculated profiles are sensitive should be perturbed in a sensitivity analysis, and comparisons of calculated and observed water quality obtained for all conditions associated with available data. If the comparisons obtained in the sensitivity analysis between observed and calculated water quality "deteriorate" relatively uniformly in a biased manner, using the perturbed coefficient values, this is usually evidence that the diversity of conditions under which the model has been tested is large enough to have defined a reasonable value for the coefficient. If the comparisons do not deteriorate and the waste allocations change, then data and calculations under different conditions of flow, temperature, and load should be obtained to assist in defining the reaction coefficient. In certain situations special field or laboratory studies, such as reaeration measurements or bottom release rates of oxidizable substances, may also be of value in defining the coefficient.

Model verification efforts should contain activities similar to those discussed under model calibration. In general, reaction coefficients and class II sources should not be altered in the verification analysis. If changes in these model inputs are required, the changes should be entered for all data sets including historical, calibration, and verification data. Only one set of reaction coefficients and category II sources is appropriate unless documented cause and effect variations can be defined. The Category I sources are measured with each water quality data set.

Comparison between observed and calculated water quality for all data sets should be developed. The measures of adequacy discussed under model calibration are also appropriate for use in the verification activities.

The next step in the process is to define a site-specific water quality model that consists of:

- A single set of model parameters that were developed and used in the calibration and verification analysis. These parameters should be uniform in space and time varying only as defined below.
- A set of rules for variation of model parameters in terms of measured information, such as temperature, flow, loads, geometry, etc. The rules for variations of parameters should be those used in the calibration and verification activities.
- A range of values for model parameters that cannot be adequately defined by a single value. The range of parameters, determined from sensitivity analysis, should be used in all projections.
- The quantitative and qualitative measures of model adequacy, including graphs, statistics and appropriate discussions. In most situations, a quantitative measure of the difference in calculated and observed profiles equivalent to "D(o-c)" in the example is also required.

Thus, the definition of a site-specific water quality model is developed in the total context of the calibration and verification analysis. The level of understanding of the factors controlling water quality is defined in terms of the assessment of model adequacy. Both strengths and weaknesses of the analysis are identified.

3.5 ALLOCATING WASTE LOADS

Objective of Waste Load Allocation

The purpose of the waste load allocation analysis is to define the quantity of waste that may be discharged into a stream or river while meeting the water quality objectives at the lowest cost. The allocation analysis has application when two or more sources of waste affect water quality. The sources may be combinations of point and nonpoint sources or exclusively point or nonpoint sources, and all or some of the sources may be controlled.

Definition of the Allocatable Load(s)

The initial requirement is to quantitatively define the critical conditions of flow, temperature, and in some instances loading (such as wet weather events) that will control waste load allocations. There may be one or more critical conditions that should be considered as discussed under the problem identification section. Traditionally, the critical stream flow is the seven-consecutive-day low flow, which is anticipated to occur on the average once every ten years. The critical temperature is usually associated with this summer low-flow period and the flow and temperature should be consistent. EPA is currently investigating the subject of design flow and temperature conditions as well as the concept of seasonal WLAs. It is expected that technical policy guidance will be issued at some future time.

The critical conditions are employed in the calibrated and verified water quality model to project critical dissolved oxygen profiles. These profiles are developed, considering the projected wasteloads from all sources. If residual uncertainties are associated with the model in terms of coefficients, projections should also be made with the full range of model coefficients. The procedures discussed in the remainder of the wasteload allocation section should be employed to examine all the projected dissolved oxygen profiles. These projected profiles should include the total dissolved oxygen and the unit dissolved oxygen deficit responses associated with each load and source.

The total dissolved oxygen profile under critical conditions is employed to define the locations in space and time where the water quality does not meet standards or creates other water quality problems. A single location in space and time associated with the minimum dissolved oxygen usually controls the allocation process. In some instances, several locations in space or time that are below standards will control the allocation. It should be observed that questions relating to the frequency of the severity and extent of violations (and the appropriateness of specific standards) may become factors to be considered at some future date. The procedures discussed should be employed for all critical locations.

The waste load allocation process considers the critical locations(s), and the saturation value at the critical temperature. The basic procedure consists of subtracting, from the saturation value, all deficit responses for sources and processes that will not be allocated and also subtracting the difference in dissolved oxygen between the model and the observed data. The water quality objective, such as a dissolved oxygen standard, is in turn subtracted from the remainder. The resulting dissolved oxygen value is allocatable. The allocatable load is calculated by:

$$La = \frac{Da}{Dp} \times Lp \qquad (3\text{-}6)$$

where: La = allocatable load (lbs BOD/day)

Da = allocatable dissolved oxygen deficit (mg/l)

Dp = projected dissolved oxygen deficit at the critical location from controllable sources (mg/l)

Lp = load from controllable sources used in the projection for critical conditions (lbs BOD/day)

Example Allocation Procedure for Single Source

Assume that a waste load allocation study has been performed in which several data sets have been analyzed by a water quality model, that the model has been calibrated and verified using procedures described earlier, and that the verification is judged to be acceptable.

Assume that the following is indicated by use of the verified model to project dissolved oxygen impacts in the river under design conditions (future point source waste load and critical stream flow and temperature conditions).

• Dissolved oxygen saturation at design temperature (28°C)..	7.8 mg/l
• Dissolved oxygen deficit at critical location from uncontrollable upstream sources	
Initial deficit	0.2 mg/l
Oxidation of BOD (2.85 mg/l)	1.0
Oxidation of NH_3 (0.15 mg/l)	0.3
TOTAL DUE TO UNCONTROLLED SOURCES	1.5
from point source waste load to be controlled	
Oxidation of BOD	3.0
TOTAL DEFICT-all sources	4.5
• Projected minimum DO concentration	3.3 mg/l

Assume further that a statistical comparison of calculated vs observed dissolved oxygen has been performed as described earlier on all data sets used in model calibration and verification, using coefficients which were selected and used in the projection (i.e., the verified model) and that a probability plot of the absolute difference between calculated and observed concentrations is as shown by Figure 3-8.

This plot can be considered to represent one measure of the level of understanding of the system response to the waste loads identified, and in addition reflects the influences of those factors that are not included in the analysis and are causing dissolved oxygen variations. It should be cautioned that, when a test of uncertainty involves comparisons between predicted and observed values, the number and arrangement of sampling stations is critical to the meaningfulness of the test.

If environmental risks associated with maintenance of a minimum DO concentration of 5.0 mg/l are considered to be sufficiently great, then the above probability distribution, which reflects a level of uncertainty in the water quality projections, may be considered as a basis for assigning a "safety factor"

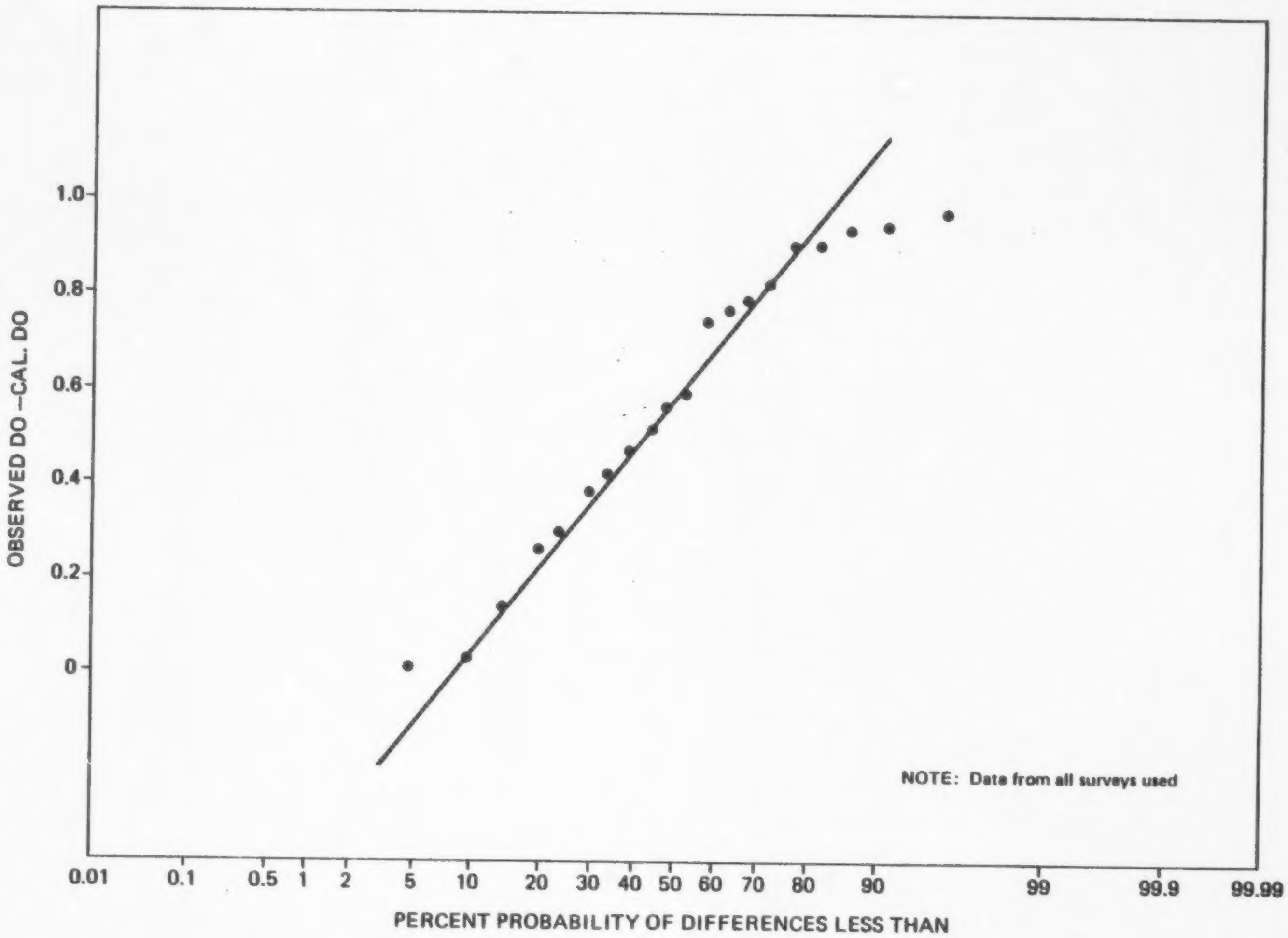


Figure 3-8. Probability of absolute difference in calculated vs. observed dissolved concentration.

in the allocation process. If one standard deviation about the mean difference between observed and calcualted concentration were adopted as the basis for selecting a safety factor as a hedge against residual uncertainty, then the 84% probability value represents the dissolved oxygen deficit that would be selected (84% of the values are less than the mean difference + 1σ). This criterion results in a deficit concentration of 1.0 mg/l. Similarly, if the mean difference were used as the criterion, a "safety factor" of 0.6 mg/l would be selected.

There are no "rules" that can be assigned to suggest what a reasonable criterion should be in a particular case. Individual judgment would be required, taking into account the environmental risk associated with the residual uncertainity as well as other uncertainties in population growth, upstream changes, etc., and reserve policies. However, the procedure should assist in forming such judgments by providing a degree of quantification to the issue.

Allocation of waste load for BOD to the point source to be controlled can then be summarized as follows:

SATURATION CONCENTRATION	7.8
TOTAL DEFICIT AVAILABLE	7.8
DO STANDARD	-5.0
DEFICIT AVAILABLE w/o VIOLATION	2.8
DEFICIT USED BY UNCONTROLLABLE LOADS	-1.5
DEFICIT THAT CAN BE ALLOCATED	1.3
LESS RESERVE FOR UNCERTAINTY IN PROJECTION OF WASTE LOAD IMPACTS (USE MEAN 50% DIFFERENCE)	-0.6
LESS RESERVE FOR FUTURE GROWTH (PER REGULATORY AGENCY POLICY)	-0.4
DEFICIT TO BE ALLOCATED TO POINT SOURCE BEING CONTROLLED	0.3 mg/l

The maximum waste load (BOD) that can be discharged, without exceeding the DO deficit that has been allocated to the point source in question, can be readily calculated by the ratio to the deficit caused by the existing load. Thus, under critical design conditions:

DO Deficit for 4000#/day BOD =3.0 mg/l

$$\text{Allowable waste load} = \frac{0.3}{3.0} \times 4000\ \#/\text{day} = 400\ \#/\text{day}$$

In this case, the waste load allocation for the point source in question would be 400 #/day; a treatment level that provides 90% reduction would be required.

It is convenient to observe at this point that "safety factors" developed in the above manner, based essentially on residual uncertainties in the model projections of impacts, will in some cases preempt a significant portion of an otherwise allocatable load. The type of residual uncertainties in projected impacts addressed by the indicated probability analysis will tend to be greater when data and data acquisition are limited, and model verification efforts are constrained as a result. Where the economic impact of providing treatment is substantially influenced by the magnitude of such an assigned safety factor, and if environmental risks do not justify neglecting this consideration, then additional model verification efforts (with attendant data acquisition) may be appropriate.

Waste Load Allocations for Multiple Sources

When two or more sources are subject to application of controls, considerations become more complicated. Two types of situations can be identified to define the required details of the waste load allocation analysis.

In the first instance, removals of load from any source yields a comparable water quality response. A comparable load response may be encountered when:

- All sources are geographically close to each other as defined by the receiving water response.
- Reaction rates for different sources vary so that maximum impacts (such as maximum dissolved oxygen deficits) occur at the same location.
- The contaminant controlling water quality is conservative or slowly reactive so that impacts accumulate downstream. An illustration of this latter situation might be nutrients that are not used by phytoplankton until stream geometry changes downstream, or NH_3 discharges that do not nitrify until a suitable downstream environment is reached.

The second situation is characterized by spatial or temporal variations in the water quality response from various sources. Variable load response may be encountered when:

- Sources are displaced geographically so that the contributions of each load at a critical location are different and not the maximum response to the load.
- The timing of loading inputs is different for each load so that water quality is controlled by different sources depending on time. Illustrations of this situation are associated with the short-term impacts of wet weather loads from CSO and urban runoff in contrast to point source loads.

Variable load response can lead to considerations such as: probability of simultaneous peak loadings and treatment of a group of dischargers as a "cluster". Cluster analyses are the subject of future guidance documents. It is hoped individual allocations can be coordinated so that loading restrictions' impact, on the dischargers, is less than if considered individually, without compromising the water quality objectives.

The following discussion addresses the comparable load response and variable load response allocation procedures. The waste load allocation analysis must define which situation exists for a site-specific project.

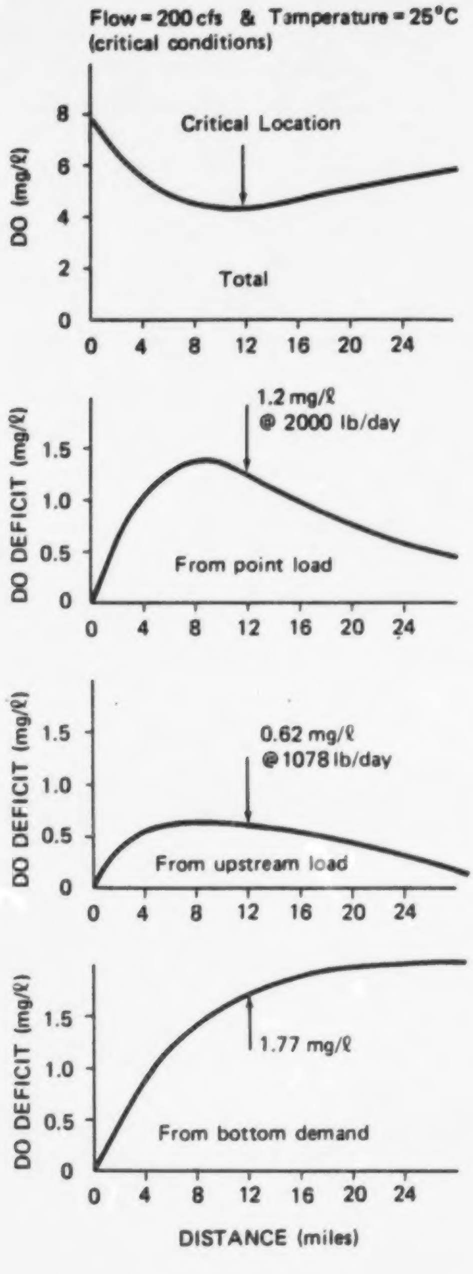
Unit response curves developed in the calibration and verification analysis will provide the necessary information if one or more calibration calculations were obtained near critical conditions. If not, a series of unit responses at critical design conditions will be required. If all controllable loads have essentially parallel deficit responses at critical conditions, then the comparable load response analysis procedure can be employed. If all controllable load responses are not essentially parallel at critical conditions, then the variable load response analysis procedure must be used.

The following example is presented to illustrate a comparable load response situation. The variation in procedure for variable load response situations is discussed following this example.

Figure 3-9 contains a numerical example of the details of the calculation procedure for the allocatable waste load. It is assumed that the critical conditions are 200 cfs and 25°C, the loads and responses are shown, and that both the point source and the load from the upstream source are controllable. The critical location for dissolved oxygen is in the region of mile point 12, and the allocatable dissolved oxygen deficit is 1.07 mg/l, with the associated allocatable load equal to 1808 lbs/day.

A formal method of defining comparable load response and variable load response situations is to compare the ratios of the mass discharge required to produce one mg/l of dissolved oxygen deficit response for each source at the critical location. For the example problem, these load ratios are 1667 and 1738 lbs/day/mg/l for the point and nonpoint sources respectively. Therefore, removal of one pound of BOD from each source yields essentially the same water quality improvement.

In the foregoing situation, removal of a pound of contaminant, such as BOD_5, from any source yields a comparable water quality response. In the



	mg/ℓ DO
Saturation Concentration	8.24
Total Deficit Available	8.24
DO Standard	−5.00
Deficit Available w/o Violation	3.24
Deficit Due to Uncontrollable Load (Sediment O_2 Demand)	−1.77
Deficit Which Can Be Allocated	1.47
Less Reserve for Uncertainty in Projection of Waste Load Impacts (from statistical analysis of differences of calculation vs observation)	−0.40
Deficit to be Allocated Among Controllable Sources (point source and upstream load)	1.07

$$L_A = \frac{D_a}{D_p} \times L_p$$

$$L_A = \frac{1.07}{(1.2 + 0.62)} \times (2000 + 1078)$$

L_A = 1808 lb/day
(allocatable waste load)

Figure 3-9. Example of the calculation procedure for allocatable load.

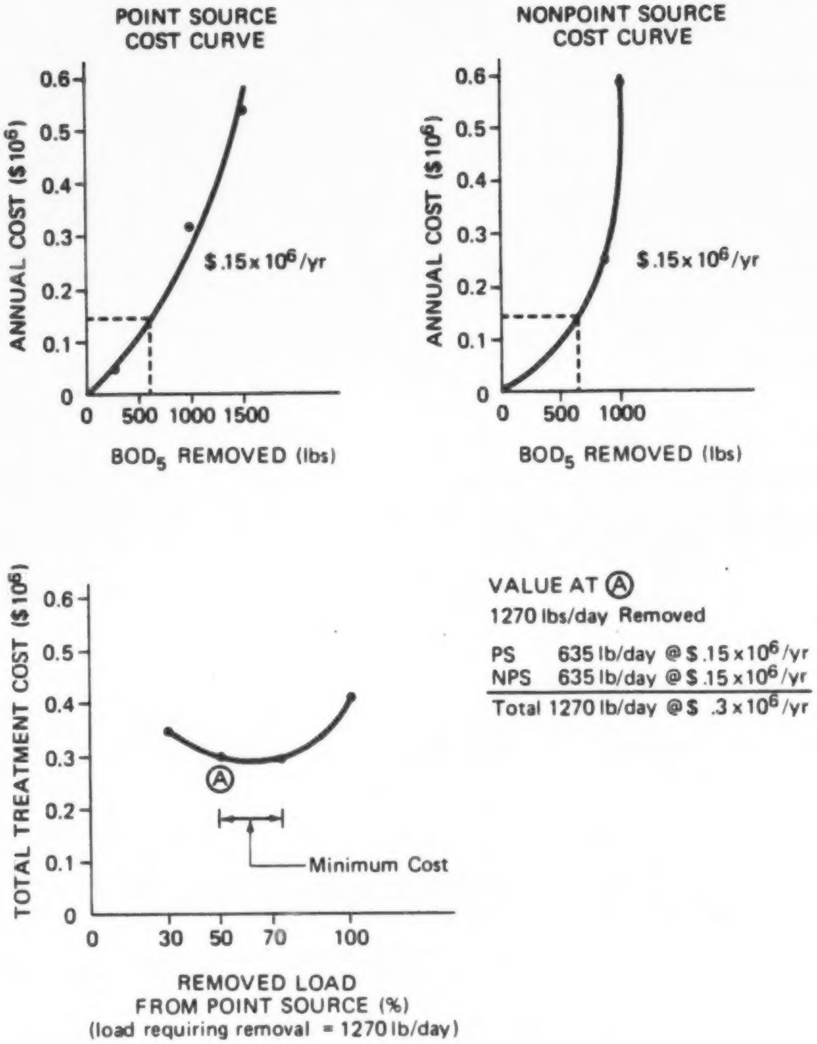
example the allocatable load is 1808 lbs/day, which requires a total removal of (3078 lbs/day - 1808 lbs/day) 1270 lbs per day.

The next issue is that of allocating this load between the sources subject to control--in this case the point source and the source that is contributing to what is identified in this example as a residual load entering from upstream of the specific study area. Various rationales have been considered for resolving such issues, and state or local policy determinations will play a dominant role in decisions regarding distribution of allocations among separate sources. Potential approaches to this issue are discussed elsewhere in the series of guidance manuals on waste load allocations.

Recognizing that solutions that consider only technical issues will not apply in many cases, an approach to allocation of several loads, based on the above example, is presented as one possible approach.

The allocation procedure consists of defining the allocatable load as illustrated and the site-specific cost curves for treatment of each individual loading source. These curves can then be combined to define a minimum cost.

Figure 3-10 presents an illustration for the conditions examined on Figure 3-9 in the example. The cost curve for the point source is shown as a continuous function. In practice, this might well be piecewise and discontinuous. As an example, additional BOD removals from a secondary plant of approximately 500 lbs/day might be associated with gravity filtration; removals of 1000 lbs/day would probably require chemical coagulation and gravity filtration, while removals of approximately 1500 lbs/day might require these unit operations plus activated carbon. Similar considerations would be appropriate for the nonpoint source



VALUE AT Ⓐ
1270 lbs/day Removed

PS	635 lb/day @ $ .15 x 10^6/yr
NPS	635 lb/day @ $ .15 x 10^6/yr
Total	1270 lb/day @ $ .3 x 10^6/yr

Figure 3-10. Example of allocation procedure to minimize cost.

curve. This curve might also have a higher degree of uncertainty associated with both annual costs and performance in terms of BOD removal.

The specific calculation procedure is illustrated for the value at "A" where the load requiring removal is defined by:

$$L_R = L_P - L_A \tag{3-7}$$

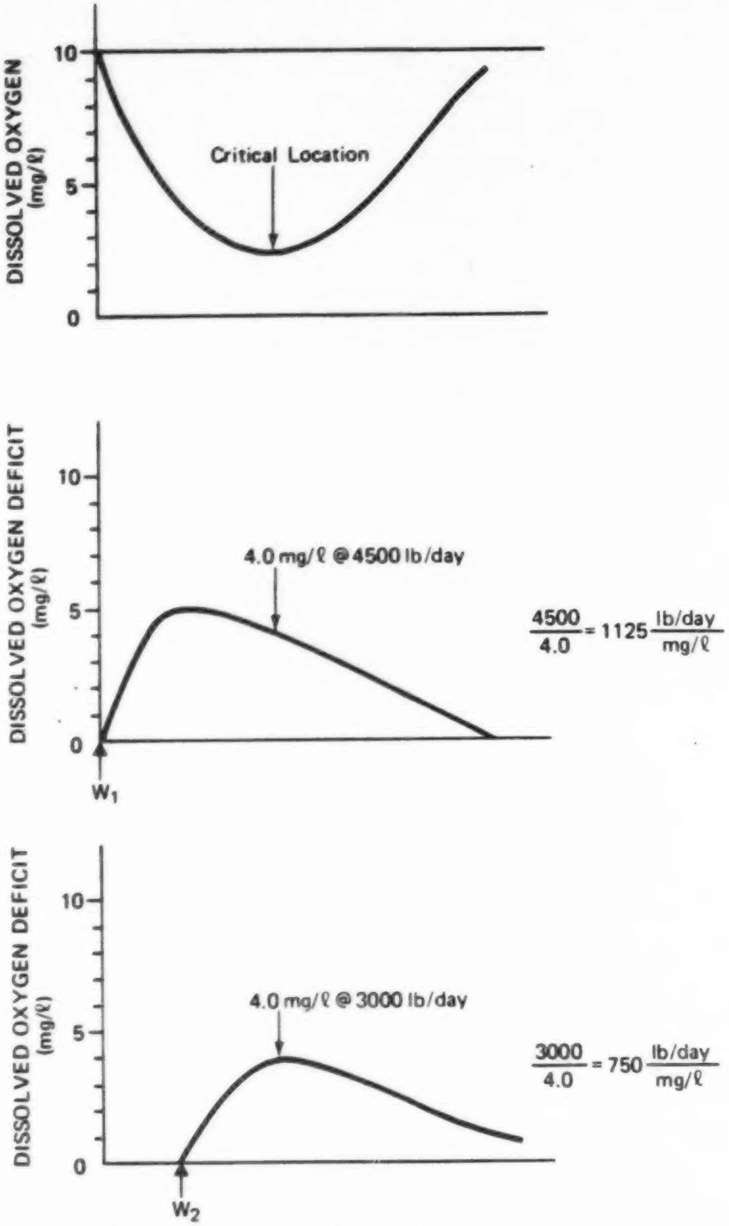
$$1270 \text{ lbs/day} = 3078 - 1808$$

Where: L_R = load requiring removal (lbs/day)
L_A = allocatable load (lbs/day)
L_P = projected load from controllable sources

The value at "A" is for 635 lbs/day removal from both the point source and non-point source as shown. Removals and costs are then summed for both loads. A comparable procedure can be used for more than two loads; the minimum cost region is shown. This region should be checked to confirm that treatment design is feasible at the removals required. The allocated load to each source is the projected load of the sources minus the required removal.

Allocation Procedure for Variable Load Response

The case of a variable load response is characterized by unequal effects from removal of a pound of BOD, considering two or more sources. Figure 3-11 presents an illustration of the types of unit responses that could be found. The critical location is shown. Load W_1, which is 4500 lbs/day, contributes a deficit of 4 mg/l at this location. The deficit caused by W_2, which is 3000 lbs/day, is also 4 mg/l at this location. Therefore, one mg/l of dissolved oxygen deficit at the critical location could be obtained by removal of 1125 lbs/day of load from source W_1 or 750 lbs/day of load from source W_2. The relative effectiveness of load removals in terms of load W_2 is:



$$\frac{4500}{4.0} = 1125 \frac{\text{lb/day}}{\text{mg/ℓ}}$$

$$\frac{3000}{4.0} = 750 \frac{\text{lb/day}}{\text{mg/ℓ}}$$

$$\frac{W_1}{W_2} = \text{Relative Effect} = \frac{1125}{750} = 1.5 \frac{W_1\,\text{lbs}}{W_2\,\text{lbs}} \text{ for Same Effect}$$

If DO must be increased by 2.0 mg/ℓ there is a need to remove 2250 lbs from W_1 or 1500 lbs from W_2.

Figure 3-11. Example of variable load response system.

Relative Effectiveness (W_2) =

$$\frac{\dfrac{1125 \text{ lbs/day } (W_1)}{\text{mg/l}}}{\dfrac{750 \text{ lbs/day } (W_2)}{\text{mg/l}}} = \frac{1.5 \text{ lbs/day } (W_1)}{\text{lbs/day } (W_2)}$$

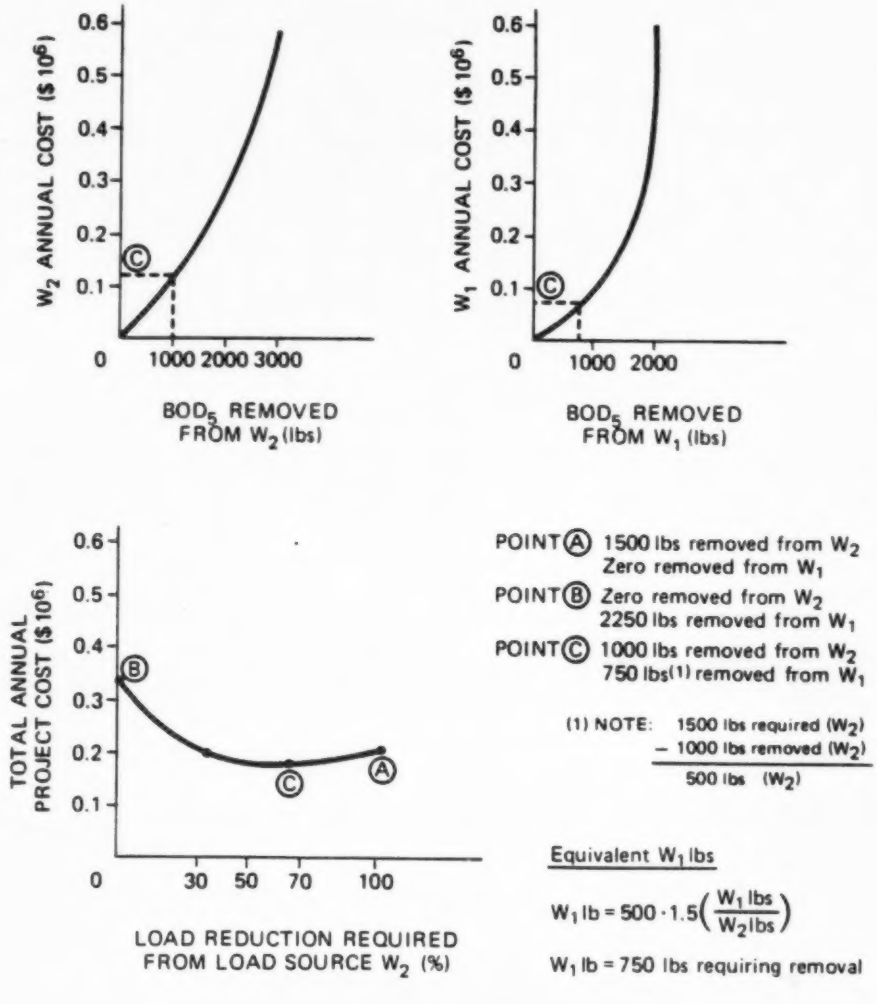
Thus, to obtain the same improvement in water quality, fifty percent more load must be removed at load W_1 as compared to load W_2. Cost curves are presented in Figure 3-12 for each source. Calculations are also presented on this figure for the total cost curve. In particular, for point (c) on the curve, an illustration is presented for removal of 1000 lbs/day of W_2 with 500 lbs/day of W_2 equivalent load requiring removal from W_1. Since the relative effectiveness is 1.5, the actual removal of W_1 load must be 750 lbs/day as indicated.

It is necessary to determine if treatment at the lowest cost point is feasible from the technical, engineering, and economic standpoint. With a cost curve shape as flat as that shown in the figure, there is a wide latitude in selection of the actual design.

If the waste load allocation procedures stop short of the cost evaluation step, the relative effective ratio can be employed to define allocations between sources. In particular, take as an example the calculations for point (c) without regard to costs:

> 1500 lbs/day of load W_2 equivalent removal is required overall as shown by the assumptions on Figure 3-11.
>
> Assume 1000 lbs/day of W_2 removal is to be provided.

The W_1 load that must be removed is (1500 - 1000) lbs/day = 500 lbs/day as equivalent (W_2) load. Since the effectiveness ratio is 1.5, the W_1



POINT Ⓐ 1500 lbs removed from W_2
Zero removed from W_1

POINT Ⓑ Zero removed from W_2
2250 lbs removed from W_1

POINT Ⓒ 1000 lbs removed from W_2
750 lbs[1] removed from W_1

(1) NOTE: 1500 lbs required (W_2)
− 1000 lbs removed (W_2)
500 lbs (W_2)

Equivalent W_1 lbs

$$W_1\,\text{lb} = 500 \cdot 1.5\left(\frac{W_1\,\text{lbs}}{W_2\,\text{lbs}}\right)$$

W_1 lb = 750 lbs requiring removal

Figure 3-12. Example of allocations for variable load response system.

removal required is:

$$W_1 \text{ removed} = 500 \text{ lbs/day as equivalent } (W_2) \text{ load X}$$

$$\frac{1.5 \text{ lbs/day } (W_1)}{\text{lbs/day } (W_2)} = 750 \text{ lbs/day } (W_1)$$

and the allocations for W_1 and W_2 are:

$$3750 \text{ lbs/day } (W_1) = 4500 \text{ lbs/day } (W_1) - 750 \text{ lbs/day } (W_1)$$
$$2000 \text{ lbs/day } (W_2) = 3000 \text{ lbs/day } (W_1) - 1000 \text{ lbs/day } (W_2)$$

The procedures indicated above would have to be applied for all loads and at any critical locations or times as required in a site-specific analysis.

SECTION 4.0

TECHNICAL CONSIDERATIONS

4.1 WATER QUALITY PROBLEM DEFINITION

General

A discussion of the water quality problem to be addressed by the WLA will serve a number of useful purposes. For officials or agencies responsible for reviewing the WLA study and with making or approving decisions, the problem discussion can provide a clear summary of the nature, extent, and significance of the problem, and thereby provide a basis for comparing the changes to be achieved by adopting the WLA. An effective problem definition can also help the technical staff performing the WLA study to keep the important issues in focus during the progress of the effort, to determine the most appropriate modeling approach and data needs, and to evaluate the significance of uncertainties in the modeling analysis.

In this discussion, it is assumed that the broader aspects of the problem identification tasks (discussed in Book I) have resulted in a determination that BOD/DO impacts will be the focus of the WLA. The additional detail necessary to define the pertinent aspects of such problems is discussed here.

"Problem definition" is a task that persists throughout the WLA effort and is refined and updated at different points during the program. This orientation will help to maintain focus on the underlying objective: to develop a sufficient understanding of the receiving water system and its responses to waste loads, which will permit sound decisions on WLAs and appropriate levels of treatment. The early tasks in a phased problem definition effort will help to direct subsequent work elements. The final product of this task becomes: (1) a clear description of the "problem"—its nature, magnitude, and spatial and temporal characteristics; (2) an indication of how the waste load being allocated is related to the problem; and (3) a description of how the allocation selected will modify the problem.

Problem Definition Phases

Several phases will be involved in problem definition at increasing levels of refinement and detail, each involving attention at different stages of the WLA effort and drawing on information developed during prior tasks.

Four phases are outlined below. The number of phases may vary depending on the local situation; therefore, the summary should be considered illustrative of several stages in a single process. Results from each of the phases will be aggregated and consolidated for the final output of the problem definition task.

Initial Phase. The principal purpose is to direct and focus the WLA study. Principal tasks involved in initial problem definition include the acquisition and analysis of historical data, i.e., water quality, stream flow, and waste loads, and possibly some simple impact analysis or dilution calculations. Any "qualitative" expressions of a problem (other than through numerical water quality data) would also be recorded.

The initial problem statement should also identify those factors other than water quality that will enter into the WLA process. Projected changes in population, and in development of the area, which would result in increases in raw waste loads or the introduction of new sources of load, should be identified and the level of uncertainty associated with such projections discussed. Also, any policy or policy alternatives relating to reserve allocations should be identified. A preliminary indication of the future date when waste loads are proposed to be allocated should be made.

The product of this effort would be the identification of the type of water quality problem to be addressed by the WLA (for purposes of this chapter, BOD/DO problems are selected). Tabulations or plots of data and other pertinent descriptions would be furnished in support of the determination. Population or industrial growth projections, and existing or potential policies on reserve allocations would be discussed.

Preliminary Phase. The purpose of this phase is to define important details that will influence certain aspects of the monitoring and modeling effort. The principal tasks would include a preliminary impact assessment similar to that described in the Areawide Assessment Procedures Manual, Chapter 2 (29) and waste load projections. Available data and estimates or assumptions would be used where necessary, as would the results and observations from a reconnaissance survey.

The preliminary assessment should also examine the sensitivity to uncertainties in population growth, allowances for introducing new load sources to the area, or to alternate decisions on the amount of allocatable loads that will be held in reserve.

The product would include the following types of information. The spatial scale of the impacts would be defined, so that monitoring station selection will provide adequate coverage. The relative significance of impacts under different flow or temperature regimes and the relation of current flows to future drought flows to be selected as a design basis will provide information on most appropriate times for conducting monitoring surveys. Other factors of potential importance should be identified, for example, other significant load sources, benthal deposits, algal activity, etc.

Problem definition at this stage should provide confirmation or modification of the initial assessment and furnish additional detail. Results should be used to help ensure that the monitoring program is structured effectively and that the model selected can address all the load sources and influences of importance. Results would also be used to determine the need for refined growth projections or discussions relating to reserve allocation policies.

Interim Phase. During this phase, problem definition becomes a refinement of previously developed problem statements. It would draw on data developed from monitoring efforts performed to that point, and principally on any insights or understanding of the system and responses derived from model calibration efforts. Examples of the type of refinement that might be developed at this stage would be a modified evaluation of the relative significance of a point source load in relation to sediment oxygen demands, upstream boundary loads, or the significance of diurnal DO fluctuations due to algae on violations of DO standards.

Where appropriate, the results would include a clearer appreciation of the relative significance of the load to be controlled on the water quality problem, and possibly the identification of specific monitoring tasks to provide information on important other sources (e.g., diurnal DO measurements or benthal O_2 uptake).

Final Phase. The final product of this series of tasks would be a description of the "problem," supported by the pertinent data and analysis results developed from historical records and WLA efforts. It should describe the conditions (present and future) under which specified impacts can be expected to occur, quantify the range of uncertainty based on calibration/verification of the model, and identify the changes in water quality impacts to be expected by the recommended WLA. It should further describe the effect of alternative growth levels, and/or reserve allocation policies on the recommended WLA.

Specific Steps in Problem Definition

The following tasks should be included in the development of a definition of the problem being addressed by the WLA and the detailed characterization of its important aspects. Several iterations, following the phases approach described, should be performed.

Beneficial Use. Identify the beneficial use or uses designated for the affected stream segments. This is often associated in a general way with the stream segments' formal "classification;" however, for the purposes of a WLA study, a more specific description of stream use, either existing or intended, should be made to the extent possible. Examples of such additional detail would include actual use by the public, location of recreational or water intake sites, etc.

It is important to go beyond a simple recitation of a list of "uses" associated with a particular stream classification because decisions that have some uncertainty must address the associated risk. The nature of the actual use, the local importance assigned to it, and the effect (permanent or temporary) all affect the risk and the decisions regarding load allocation. The value of such beneficial uses may often be qualified (if not quantified), for example, by the revenues generated to a local economy from sport fishing and support activities.

Water Quality Standards. Identify the water quality standards (in this case DO concentrations) that have been established to protect the assigned beneficial uses. Describe the standards in enough detail to permit subsequent evaluation of violations. For example, DO standards may be expressed as any one or all of the following:

- mean concentration
- minimum concentration
- % saturation
- statistical--e.g., % time a specific concentration may occur

Analyze Historical Data. Review, summarize, and interpret available historical data, and compare with water quality standards. The data of importance include stream flow records or estimates for the segments of concern derived from nearby gages; stream temperature records; waste load levels for the discharge in question and for other relevant sources; and operational procedures for regulated streams.

Water quality data should be secured from STORET, from reports or studies, or other available sources. It is generally best to sort the record in ways that will assist interpretation, for example:

- Spatially. Stations upstream and downstream of the discharge point, with adequate station coverage for the spatial scale of BOD/DO reactions. In preliminary evaluations, consolidation of records from nearby stations may assist and simplify initial screening.

- By flow regime. Where record can be sorted between periods of high and low flows, interpretation will be aided.
- By temperature, wet vs. dry periods, etc. to the extent possible by the nature of the available data.

Where data are in STORET, the use of several retrievals, rather than a single one, can assist in performing the above type sorting.

Preliminary Impact Analysis. Perform preliminary screening analysis using information on loads, flows, temperature, etc. developed from historical data review, and preliminary estimates using a range of coefficients for sensitivity tests. Compare preliminary projections under existing conditions with historical data to identify the extent to which the waste load in question is likely to account for the observed water quality impacts.

Using population and growth projections, and critical stream flow/temperature condition(s) being considered for use in WLA decisions, make preliminary projections of impacts expected under such conditions.

Reconnaissance Survey. During a reconnaissance survey of site sampling stations for the intensive survey program, look for evidence of other influences that could be potentially significant in a DO analysis. For example, is there evidence or reason to believe that sediment oxygen demand might be significant in some reaches? Are there any obvious NPS loads or significant tributary inflows that might affect water quality during dry weather or wet weather, when such conditions cannot be sorted from the historical data set?

Define Problem. From the analysis of historical data, the preliminary impact analysis, the reconnaissance survey and other available evidence, develop as detailed a description of the "problem" as is possible at that stage of the study. In addition, identify the factors that should be addressed or refined further in subsequent program tasks. These could include refined estimates of future load projections, other pertinent load sources to be covered by the monitoring program (e.g., sediment oxygen demand), tributary flow and quality data, etc.

Several levels of problem description should be incorporated in the problem definition because the distinctions may be important in subsequent judgments regarding the WLA. They may further assist in prioritizing future actions among a number of separate WLAs.

- Public perception and concern. Is there a general public recognition of a problem, and an active interest in its resolution (fish kills, closed beaches, etc.).
- Actual use impaired. An active current use assigned to the stream is being denied, impaired or threatened (recreational fishery being degraded, restricted use of beaches, etc.).
- Violations of standards. Water quality standards associated with the water segment classification are violated, or projected to be violated. Violations may be indicated by historical data, projected to occur based on preliminary screening, or projected to be possible by preliminary screening and confirmed by model calibration/verification.

The problem should be described in as much detail as possible, including discussion of the magnitude, location and frequency, and, to the extent possible, the conditions under which the problem will occur.

4.2 DATA REQUIREMENTS

Introduction

The type and amount of data available for performing a waste load allocation will determine the confidence that can be placed in projections of the effects of different decision alternatives. Depending on the adequacy of available historical data and the risk (environmental and economic) associated with uncertainty in projected impacts due to limitations in historical data, one or more intensive surveys may be required. It is preferable to consider which type of model will be used in the WLA at the earliest stage of the data collection effort, since the choice of model will affect data needs.

The objective of data acquisition in a WLA study, whether from a search of historical records or from additional field surveys, is twofold.

- To provide a sound basis for assigning rate coefficients and other critical input values, such as stream geometry, loads, etc., which relate to the specific site being examined.
- To contribute to the development of an understanding of how the receiving water system responds to various load levels and sources. This is achieved through the use of data in the calibration and verification of water quality models.

The methodology described in this manual recommends that an initial or preliminary impact assessment be performed using available historical data. Among the benefits of such a calculation is the information it will provide to assist in structuring intensive surveys or other data acquisition activities for the WLA study. The collection of new data should emphasize those areas where the greatest uncertainty exists, and where WLA decisions are sensitive to the values selected. Examples of such information include the following:

- waste loads (sources) that should be monitored
- the flow regimes at which transport studies are required
- the most appropriate conditions under which intensive stream surveys should be performed (waste loads, stream flow, temperature)

- the spatial coverage that should be considered for station location
- special studies that may be required

Methodologies for performing such initial assessments are described in some depth in both the Areawide Assessment Procedures Manual (AAPM, 29), and in Water Quality Assessment: A Screening Method for Non Designated 208 Areas (38). In addition to providing a preliminary screening to aid in the design of an effective monitoring program, these methodologies can also be used to provide estimates for Category II loads, which will not be monitored.

In addition, Chapter 2 of the AAPM provides information on sources of historical data that should be searched, as well as instructions for access to some sources. Appendix D of this document presents general guidelines on station location (Part I). It also includes a Parameter Handbook (Part II) that presents a concise summary of enough salient information about each water parameter to aid in decisions concerning the likelihood of the constituent's presence in a particular stream or discharge and its effects on water quality or use. It discusses factors pertaining to sampling and analysis of the constituent that should be considered in determining the ramifications of including the parameter in a water quality monitoring program. It also presents information on analytical methodology, including sample quantity and preservation and handling considerations.

The material that follows in this section presents more specific guidance and recommendations pertaining to data requirements for WLA that address BOD/DO problems in rivers and streams. It should be evaluated in conjunction with the broader information and guidance referenced above.

Sample Station Location

Spatial plots of historical or calculated water quality should be developed by the initial assessment for a set of flow regimes of interest. One of the conditions analyzed in this manner should be current loading conditions and "typical" stream flows (i.e., those flows expected to prevail during proposed intensive surveys). These plots, together with observations made during a reconnaissance survey, should provide the basis for establishing the location of field sampling stations. The number and location of sampling stations will be controlled by site-specific factors, including access to the stream and other logistic issues as well as the cost of sampling. Normally, a minimum of four sample stations should be considered with maximum distance between stations on the order of one-half to one day's travel of water time. This represents a minimum requirement, applying to the simplest of WLA stations; for example, a single discharge to a perennial flowing stream, where the ${}_7Q_{10}$ flow is greater than the discharge flow, and there are no significant tributaries downstream. Figure 4-1 illustrates three situations that contain typical issues encountered when defining the location of sampling stations.

In illustration I, there is a single waste discharge station with no tributaries. This is a typical situation when the minimum of four sampling stations is usually adequate. Sampling station A is employed to define upstream water quality while stations B and C define the dissolved oxygen response to the load. Station D provides information in the recovery zone, which is dominated by reaeration, and may be examined for nitrification and/or increased diurnal dissolved oxygen variations.

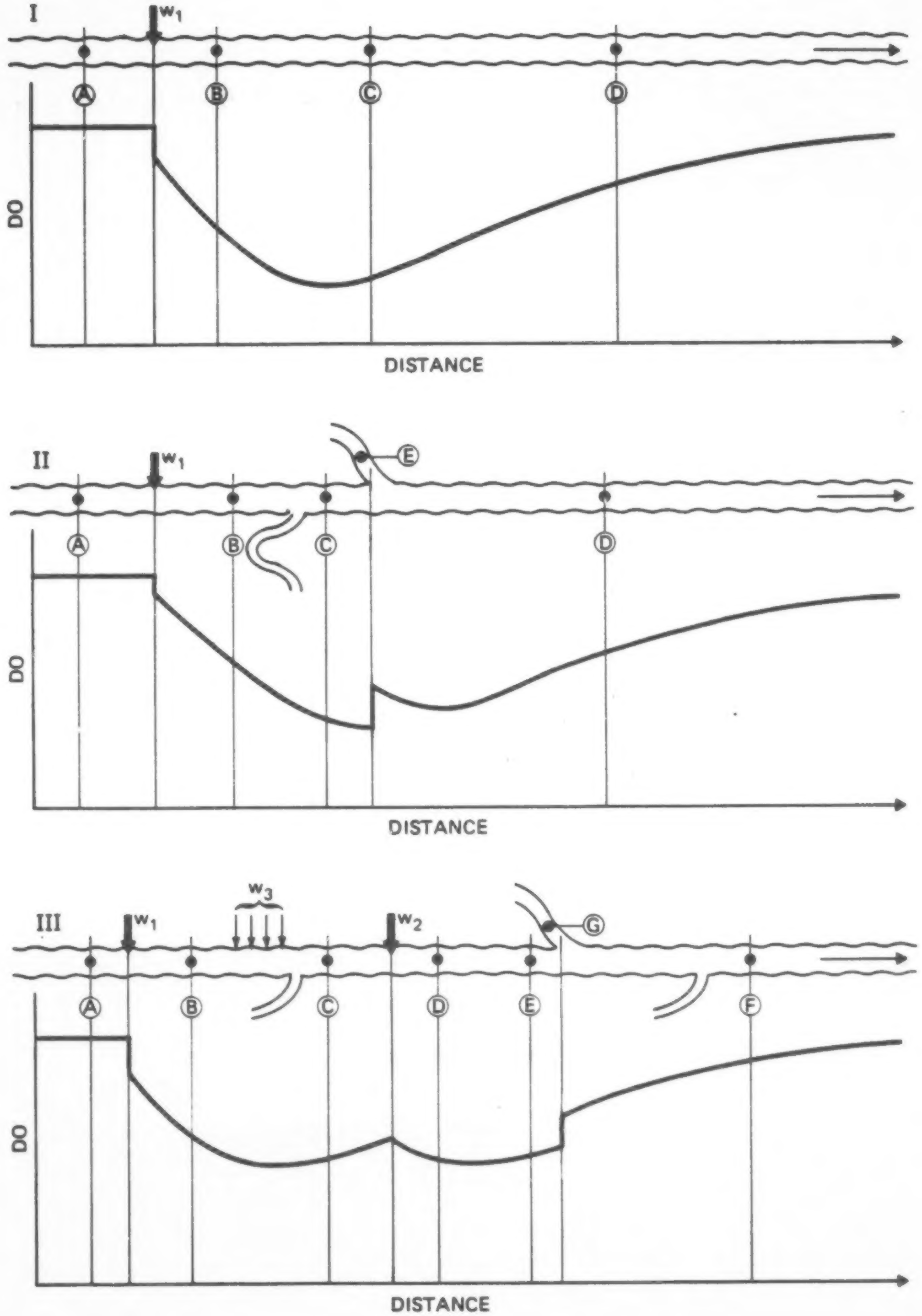

I
w_1
A
B
C
D
DO
DISTANCE
II
w_1
E
A
B
C
D
DO
DISTANCE
III
w_3
w_1
w_2
G
A
B
C
D
E
F
DO
DISTANCE

Figure 4-1. Sampling station locations.

In illustration II, several tributaries are shown to enter the system downstream of the discharge. Sampling stations A through D serve the same functions as indicated in illustration I. One additional sampling station is located on the major tributary. Minor tributaries need not be sampled. In this instance, minor tributaries are defined in a manner similar to category II load sources, i.e., they have a small influence on water quality; and errors in assigning the quality in the tributary will not influence the waste load allocation. A sensitivity calculation conducted in the initial assessment analysis could provide the information necessary to distinguish between major and minor tributaries. The category I source "W_1" and major tributaries should be sampled before and during each survey in both illustrations I and II.

Significant sources of waste load (i.e., category I loads) are sampled one or more days in advance of the instream monitoring during an intensive survey in order to ensure that, given the time of travel to the most downstream station, an accurate estimate of the waste load that affects the stream sample is available. In a true steady-state condition (i.e., constant waste loading rate and constant stream flow), this would not be important since any single measurement would represent all conditions during a steady-state period. Normally, however, the conditions selected will not reflect a true steady-state condition, but rather an acceptable approximation of one. The greater the likelihood of appreciable fluctuations, either in waste loading or more probably in stream flow, the greater the attention that should be given to lagging monitoring of waste loads and stream samples. In addition, where the likelihood of such fluctuations is high, increases in both the duration of an intensive survey and the number of samples collected per day for averaging, in excess of those recommended later in this section, should be considered.

Illustration III presents the added complexity of multiple waste sources entering the system. In this instance, sources "W_1" and "W_2" are category I sources and should be sampled, while "W_3" is assumed to be a category II source and need not be sampled during each survey. It should also be noted that sampling stations B through E are located to provide data on the overall water quality response of the stream or river and are not located adjacent to and immediately downstream of individual inputs, such as sources or major tributaries. The objective of the water sampling program is to define overall system response to waste loads. Apart from the fact that a significantly larger number of sampling stations would be called for in such a case, it is a very unique situation when upstream and downstream measurements around a load or tributary can provide meaningful data on the overall dissolved oxygen response of their system. This results from the time and space scale over which dissolved oxygen changes tend to occur. Where the appropriate shape of the dissolved oxygen sag curve, indicated by historical data or calculated in the initial assessment, is such that it covers an extended stretch of river (as illustrated here), additional sampling stations should be established along the main stem.

Water Quality Variables and Frequency of Sampling

In general, the water quality variables and the frequency of measurements obtained during a survey will be defined by site-specific water quality problems and system responses. The following comments are offered to provide some assistance in defining the details of water quality sampling programs.

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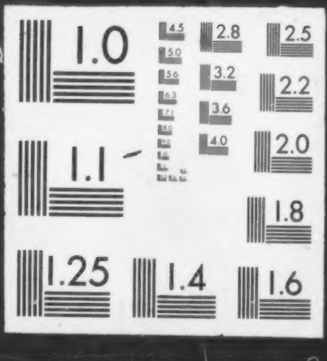
1.0
4.5
5.0
5.6
6.3
7.1
8.0
2.8
3.2
3.6
4.0
2.5
2.2
2.0
1.1
1.8
1.25
1.4
1.6

It is neither necessary nor cost-effective to analyze every variable to be measured during a water quality survey with the same frequency at every sampling station. One overall approach to defining the variables to be monitored and frequency of measurement is to structure the water quality survey in the context of the following "two key questions":

- How are the measured values of this variable to be used in detail in the waste load allocation analysis?
- Would the waste load allocation decision be expected to change if this variable were either not measured or measured less frequently or at fewer stations?

Supplemental questions that assist in this process are:

- What is the specific relationship of this variable to the water quality problem?
- What is the anticipated variability of this variable?
- What is the anticipated response time of the system to changes in value of this variable?

There are several contexts in which site-specific dissolved oxygen problems can be encountered. These are:

- Dissolved oxygen dominated by CBOD oxidation
- Dissolved oxygen dominated by oxidation of both CBOD and NBOD
- Dissolved oxygen dominated by diurnal fluctuations and oxidation of BOD.

Table 4-1 provides an indication of the minimum survey duration, number of measurements, and percent of stations that could be sampled as a function of the water quality variable and dissolved oxygen problem context. The information in this table is for a minimum effort where relatively low capital and/or environmental risks are associated with a site-specific

TABLE 4-1. SUGGESTED MINIMUM SAMPLING REQUIREMENTS

Variable(2)	DO Problem Context	Duration(1)(3) of Survey	Number of (1) Measurements	% of (1) Sampling Station
Dissolved Oxygen	All Problems	2 Days	2/Day AM/PM	100%
Temperature	All Problems	2 Days	2/Day	100%
pH	All Problems	1 Day	1/Day	100%
BOD_5	CBOD	2 Days	1/Day	50-100%
BOD_5	CBOD & NBOD	2 Days	1/Day	50%
UOD	CBOD	--	Once	50%
UOD	CBOD & NBOD	--	Once	50%
Organic-N	CBOD & NBOD	2 Days	1/Day	50-100%
NH_3	CBOD & NBOD	2 Days	1/Day	100%
NO_3	CBOD & NBOD	2 Days	1/Day	100%
NO_2	CBOD & NBOD	--	Once	25%
Flow	All Problems	2 Days	1-Day	1 Sta.
Time of Travel	All Problems	--	Once	100%
Velocity and Depth	All Problems	2 Days	1/Day	100%
Reaeration(4)	All Problems	--	Once	100%
Bottom Demand(4)	All Problems	--	Once	100%
Light & Dark Bottles	Eutrophication	1 Day	--	50%
Diurnal	Eutrophication	1 Day	--	50%
Nitrifier Counts(4)	NBOD	--	Once	50%

Notes:
1. Suggested minimums should be increased for more complex problem settings.
2. Other variables may be added, SO_4, TOC, COD, etc.
3. Source measurements add one day before survey.
4. Contingent on problem setting and available funds.

allocation. A more normal level of effort would provide twice the minimum number of measurements per day at all sample stations.

In general, our level of confidence in understanding system responses and, hence, in decisions on waste load allocations increases with the amount of data available for analysis. However, more important than the absolute number of data points are the specific contaminants included, whether they are measured concurrently, the spatial coverage, and the times and conditions under which the date were obtained.

It has been suggested that the equivalent of two independent data sets will normally be required in what has been characterized as the typical or "base case." A good set of historical data will often satisfy the requirement for one of these sets. As discussed in the manual section, which addresses "Level of Effort" (Section 1), specific circumstances may justify lesser, or require greater, amounts of data. The criteria to be applied in determining the number of data sets and the amount of data in any set would include the capital costs involved, the environmental risk associated with WLA decisions, the adequacy of model calibration and verification, and the sensitivity of the decisions to be made on the residual uncertainty in the prediction of water quality responses.

There may be site-specific justification for including measurements of additional variables such as TOC, COD, suspended solids, toxics, heavy metals, the phosphorus series, chlorophyll, SO_4, Alk, Na, Ca, etc. These can materially increase the cost of analysis and data handling/reduction. Use of the "two key questions" approach will provide some insight into the value of measuring additional variables.

4.3 QUALITY ASSURANCE FOR WASTE LOAD ALLOCATION STUDIES

As used here, quality Assurance (QA) is a system of activities used to provide documented assurance that a data product of known and acceptable quality is being produced.

The importance of QA should be evident; however, because of the additional effort it requires in advance planning, management, supervision, and resources, it is often neglected or overlooked. This manual has addressed, at some length, guidelines for the analysis of data that will lead to the performance of technically sound, defensible WLA studies. It is imperative that any data secured, on which such analyses are based, be adequate and defensible, i.e., of proven and acceptable accuracy and precision. This is particularly important where decisions having serious economic and environmental import will be based on the results of the WLA study.

A properly planned and implemented QA program will enable the substantiation of data accuracy and precision by an outside impartial review, and forestall any attempts to discredit or impeach the data that are produced. This section is intended to outline the minimum QA effort that will be required to ensure a reliable WLA study. Thus, its aim is to assist the user in developing a reliable and effective Quality Assurance program that will meet data user requirements for completeness, precision, accuracy and comparability of data. The Quality Assurance requirements are minimum and are not to be viewed as complete. They are presented rather as a foundation upon which the user can build a viable QA program.

Accuracy and Precision

Accuracy refers to agreement between the measurement and the true value of the measurand, with the discrepancy normally referred to as error. Precision refers to the reproducibility (repeatability) of the measurement, when repeated on a homogenous, time-stationary measurand, regardless of the displacement of the observed value from the true value.

The statistical measures of location or central tendency (e.g., the various averages, mean, median, mode, etc) are related to accuracy. The statistical measures of dispersion or variability (e.g., variance, standard deviation, coefficient of variation, and other measures derived from central moments of the probability density function) are related to precision.

Discrepancies between the results of repeated observations, or errors, are inherent in any measurement process, since it is recognized that the true value of an object of measurement can never be exactly established. These errors are customarily classified into two main groups: systematic and random (or accidental) errors. Systematic errors usually enter into records with the same sign and frequently with either the same magnitude (e.g., a zero offset) or with an establishable relationship between the magnitude of the measurement and the error. The methods of symmetry and substitution are frequently used to detect and quantify systematic errors. In the method of symmetry, the test is repeated in a symmetrical or reversed manner with respect to the particular condition that is suspect. In the method of substitution, the object of measurement is replaced by one of known magnitude (a calibration standard); an instrument with a known calibration curve is substituted for the measuring instrument in question, and so on. Thus, systematic errors bear heavily on the accuracy of the measurement.

Random errors, on the other hand, are due to irregular causes, too many in number and too complex in nature to allow their origin to be determined. One of their chief characteristics is that they are normally as likely to be positive as negative and, therefore, are not likely to have a great effect on the mean of a set of measurements. The chief aim of a data quality assurance effort is to account for systematic errors and thereby reduce errors to the random class, which can be treated by simple probability theory, in order to determine the most probable value of the object of observation and a measure of the confidence placed in this determination.

Elements of a QA Program

The basic elements of any quality assurance program include the following:

- Management's commitment to provide resources necessary to implement quality assurance activities (approximately ten to twenty percent of total water monitoring resources)
- Designation of a quality assurance coordinator responsible for coordinating and implementing necessary quality assurance activities
- Documentation of a quality assurance plan outlining the specifics of and responsibilities for the development and implementation of internal and external quality assurance checks

A complete QA program for water quality measurements would incorporate a variety of specific elements. These can be depicted on a Quality Assurance Wheel as shown in Figure 4-2. The wheel arrangement illustrates the nature of a quality assurance system that will address all elements and at the same time allow program managers the flexibility to emphasize those elements that are most applicable to their particular program. Quality assurance elements are grouped on the wheel according to the organizational level to which responsibility is normally assigned. These organizational levels are

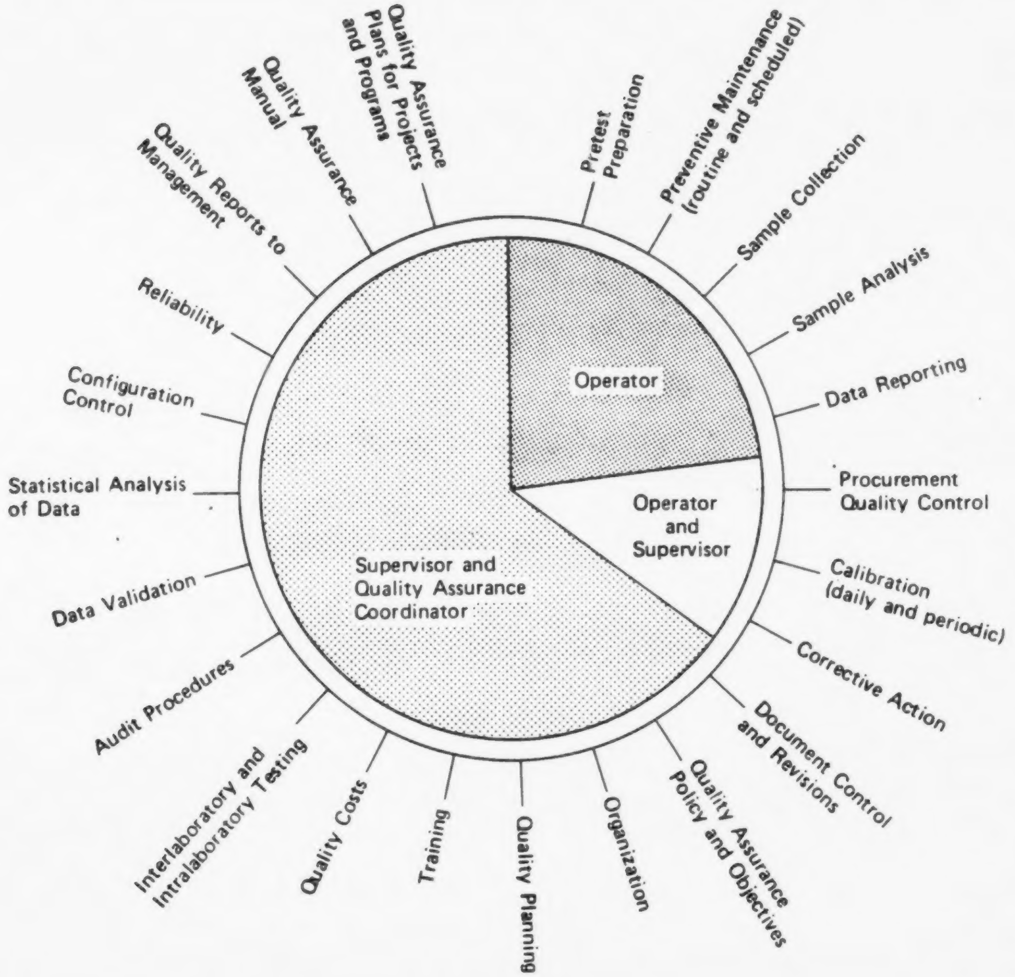
Quality Assurance Plans for Projects and Programs
Pretest Preparation
Preventive Maintenance (routine and scheduled)
Sample Collection
Sample Analysis
Data Reporting
Procurement Quality Control
Calibration (daily and periodic)
Corrective Action
Document Control and Revisions
Quality Assurance Policy and Objectives
Organization
Quality Planning
Training
Quality Costs
Interlaboratory and Intralaboratory Testing
Audit Procedures
Data Validation
Statistical Analysis of Data
Configuration Control
Reliability
Quality Reports to Management
Quality Assurance Manual
Operator
Operator and Supervisor
Supervisor and Quality Assurance Coordinator

Figure 4-2. Quality assurance elements and responsibilities (the quality assurance wheel).

the quality assurance coordinator (normally a staff function), supervisor (a line function), and operator. Together the supervisor and quality assurance coordinator must see that all these elements form a complete and integrated system and are achieving the desired program objectives.

The following specific elements are suggested to be minimal requirements for structuring a QA program for a WLA study. Any proposed program should be compared against these criteria to determine its acceptability.

- A written quality assurance plan should be prepared. It should define the oversight role of management, identify personnel responsible for the quality assurance program, and specify proper sample collection, use of approved measurement techniques, calibration standards and their verification, internal quality control practices, and appropriate data management controls.
- An estimate of costs associated with the quality assurance program in terms of percentage of overall project cost should be developed. Normally, a minimum of 10 percent of the estimated sample collection and analyses costs will be necessary for adequate quality control.
- A program for demonstration of acceptable performance through the use of audit samples should be established and utilized throughout the duration of the study.
- Provision should be made for performing on-site field and laboratory audits at the option of and on a schedule established by the the project officer. Such audits would evaluate performance and document the availability of all equipment and supplies necessary for successful execution of the study.
- Documentation of quality control performance should be submitted with the final report and otherwise as directed by the Project Officer.

Aspects of a QA Program

A number of aspects of a QA program must be addressed by the QA plan, if the minimal requirements are to be met. These aspects can be aggregated into three general categories: water chemistry (laboratory), field data collection, and data handling and reporting.

Minimum QA Requirements for Water Chemistry:

- Quality control management manual
  - Outline of Quality Assurance program objectives
  - Outline of the administrative structure of the laboratory (including an organizational chart)
  - Clear identification of the responsibilities for implementing the specific quality control activities
  - Commitment of resources by management to implement the necessary quality control activities
  - Description of laboratory training program
  - Designation of a laboratory quality assurance coordinator, including a statement addressing coordination responsibilities and duties

- Laboratory operations manual
  - Description of analytical methodologies and procedures
  - Description of laboratory quality control activities
  - Description of the laboratory's internal chain-of-custody procedures
  - Description of general laboratory requirements
  - Description of laboratory communication and coordination

- Sample log manual
- Quality control records manual
- Blind duplicate and spiked field samples
  - Sample audits
  - Parameters included in the program
  - Audit sample preparation procedures
  - Data evaluation
  - Audit follow-up and corrective action
- Estimation of limits for laboratory accuracy checks

Minimum QA Requirements for Field Data Collection:

- Sampling network design
- Sampling procedures
- Calibration of direct-reading field instruments and fixed continuous monitoring devices
- Record keeping
- Quality Assurance checks in field sampling
- Personnel training
- Flow measurements
- Records, data storage and retrieval
- Sample handling and identification procedures: chain-of-custody
- Collection of samples/field investigations

Minimum QA Requirements for Data Handling and Reporting:

- Preprinted forms and labels
- Data sheets
- Data flow
- Significant figures and rounding procedures
- Calculation checks
- Data corrections
- Data reviews
- Reasonableness and consistency checks
- Data acceptance
- Data storage and retrieval

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